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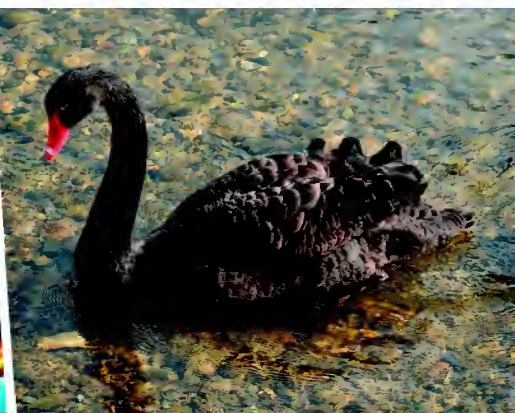
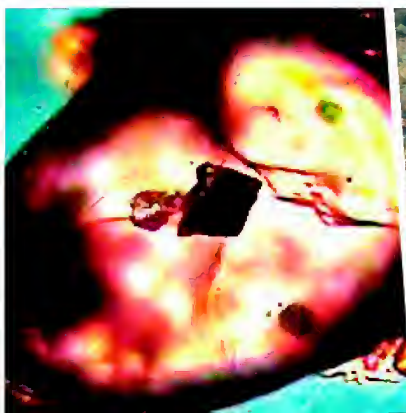
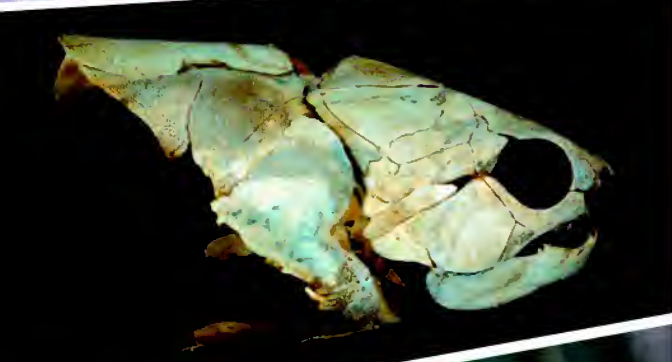
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CONTENTS

PREFACE	i
Ant (Hymenoptera: Formicidae) diversity influenced by tree thinning in the Western Australian jarrah (<i>Eucalyptus marginata</i>) forest B E Heterick, J D Majer, E D Kabay & M Loh	101
Vegetation cover and not size of remnants determines composition and diversity of ground-dwelling arthropods in native vegetation remnants J Berkelaar, P Speldwinde & C Farell	121
Ecology and Distribution of the Slider Skink, <i>Lerista neviniae</i> G Gaikhorst	131
<i>Extended Abstracts from the October 2014 Post-graduate symposium</i>	
Oral delivery of double-stranded RNA and the effects of RNA interference on the green peach aphid, <i>Myzus persicae</i> V Bilgi, J Fosu-Nyarko & M G K Jones	137
Biogas for Sub-Saharan Africa: Current situation and opportunities for improving dissemination G V Rupf, P Bahri, K de Boer & M P McHenry	139
<i>Extended Abstracts from the October 2015 Post-graduate symposium</i>	
Stratigraphic architecture and evolution of a barrier seagrass bank in the mid-late Holocene, Shark Bay, Western Australia G Bufarale & L B Collins (deceased)	141
Stabilized frequency transfer for the Square Kilometre Array D Gozzard & S Schediwy	143
Referees 2014–2015	145



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The Royal Society of Western Australia was founded in 1914. The Society promotes exchange among scientists from all fields in Western Australia through the publication of a journal, monthly meetings where interesting talks are presented by local or visiting scientists, and occasional symposia or excursions on topics of current importance. Members and guests are encouraged to attend meetings on the third Monday of every month (March–November) at 7 pm, Kings Park Board offices, Kings Park, West Perth, WA 6005, or as advertised, in the RSWA Proceedings, Diary of Events, or online.

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The *Journal of the Royal Society of Western Australia* was first published in 1914 and circulates widely within Australia and throughout the world. The journal has a large number of personal subscribers, many of whom are scientists working in Western Australia.

Cover design: The images symbolise the diversity of sciences embraced by the Royal Society of Western Australia. Counter-clockwise from the top they are: Wolfe Creek Meteorite Crater; the world-famous stromatolites at Shark Bay; the numbat (*Myrmecobius fasciatus*), Mangles' kangaroo paw (*Anigozanthos manglesii*) and Gogo fish (*Mcnamaraspis kaprios*), which are the faunal, floral and fossil emblems of Western Australia, respectively; a zircon grain (Western Australian rocks have yielded the oldest zircon dates in the world, up to 4.37 Ga); and the black swan (*Cygnus atratus*) that appears on the logos of the Royal Society and the coat of arms of the Government of Western Australia.

PREFACE

This papers in issue of the Journal of the Royal Society of Western Australia take a number of different approaches to the biodiversity of the State, and in particular the manner in which humanity has affected the communities of animals.

The first paper examines in which Ant (Hymenoptera: Formicidae) diversity is influenced by tree thinning in the Western Australian jarrah (*Eucalyptus marginata*) forest.

For several decades it has been assumed that there is a close correlation between the size of an island and its biodiversity. A number of studies had shown that this holds true both for 'real' islands - scraps of land surrounded by ocean – and 'habitat' islands. The second study in this issue examines the invertebrate communities of bushland fragments in the Albany area, in the south of the State of Western Australia. The conclusion reached is that it appears to be the extent of vegetation cover and not the size of vegetation remnants that determines composition and diversity of ground-dwelling arthropods.

The third paper is autecological rather than synecological. It documents the habitat usage, distribution and potential impacts for a little known species of skink, known only from a small area of the Cape Lambert region on the north-west coast of Western Australia. As the species is classified as vulnerable, the study is of special interest.

For some years an annual post-graduate student symposium has been run by the Society, to give an opportunity for graduate students at WA's universities to put forward their researches. Two extended abstracts from the October 2014 symposium are included here – both are on strongly 'applied' topics. One is on the oral delivery of double-stranded RNA and the effects of RNA interference on the green peach aphid, (*Myzus persicae*); the other on biogas for Sub-Saharan Africa: the current situation and opportunities for improving dissemination.

An abstract from the 2015 Symposium considers the stratigraphic architecture and the evolution of a barrier seagrass bank in the mid-late Holocene at Shark Bay. Another from the same symposium describes stabilized frequency transfer for the Square Kilometre Array, a radio-telescope now being planned for Western Australia.

Patrick Armstrong
Editor-in-Chief

December 2015

BHL



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Ant (Hymenoptera: Formicidae) diversity influenced by tree thinning in the Western Australian jarrah (*Eucalyptus marginata*) forest

B E HETERICK¹, J D MAJER^{1,4}, E D KABAY² & M LOH³

¹Department of Environment and Agriculture, Curtin University, GPO Box U1987, Perth, WA 6845

²Kabay Rehabilitation, Environmental and Biological Consultants Pty Ltd, 19 Rosebud Crt, Landsdale WA 6065

³Water Corporation, PO Box 100, Leederville WA 6902

⁴School of Plant Biology, The University of Western Australia, Nedlands, Western Australia 6009, Australia

* Corresponding author ✉ B.Heterick@curtin.edu.au

The Western Australian Water Corporation has trialled a catchment – management program in the northern jarrah forest as a possible means of enhancing water runoff and yield. Thinning has been performed by chemical injection of trees and/or by logging. This paper reports on the impact of this program on a prominent indicator component of the biota, the ant fauna. Six plots were sampled in 2006, prior to treatment and afterwards in 2008 and 2009. Two plots were unthinned controls, two were thinned, one was thinned and logged and the other was thinned, logged and burned, resulting in representations of a gradient of intervention.

However, ant abundance increased in all treated plots, especially so in the thinned, logged and burned plot. Over the three years, species richness increased in Control Plot 2, in thinned by stem injection Plot 3, in thinned by logging and burned Plot 5 and in thinned by logging Plot 6 but declined in thinned by stem injection Plot 4 and recovered somewhat but did not attain 2006 levels in Control Plot 1. Ant species evenness declined in the two thinned by logging plots. An NMDS ordination indicated that the degree of change in ant assemblage composition was lowest in the controls, increased in the thinned plots and by a further increment in the thinned by stem injection and thinned by logging plot, and was greatest in the plot that had been burned.

In summary, the various shifts in ant dynamics do not present a clear pattern over all the plots. The most noticeable changes were a shift in equilibrium between ant ecological groupings noticed in plot 3 in 2008 and the loss of cover-loving species in Plot 5 and their replacement by those that favour open ground. Plot 5, where burning was conducted, produced results that suggest an ant fauna adapted to cool, moist conditions may be in the process of being replaced by one favouring warmer conditions; with under cover nesters also replaced by open soil nesters. Potentially, there could be a loss of some species with depauperisation of the ant fauna overall. At the same time, there were no signs that the environment was being so seriously degraded that native dominants such as *Iridomyrmex chasei* (Forel) and meat ants or exotics like *Pheidole megacephala* (Fabricius) or the Argentine ant (*Linepithema humile* (Mayr)) were establishing colonies. Such species appeared to be completely absent from the survey sites.

KEYWORDS: ant diversity, functional groups, Wungong catchment, Water Corporation, tree thinning, jarrah forest

INTRODUCTION

Until very recently, the drying of the southwest of Western Australia due to climatic change has posed particular problems for people living in the Perth Basin (Barron *et al.*, 2013). The threat to the availability of potable water has to some extent been reduced by the construction of desalination plants at Kwinana and Binningup, although the increased use of recycling, dams and groundwater also continue to be important in supplying Perth's water needs (Water Corporation, 2011a). One possible way of improving the flow of water into forested catchments is by thinning of the tree cover (Bosch & Hewlett, 1982; Stoneman, 1993). However, jarrah (*Eucalyptus marginata* Donn ex Sm) forest also has a significant groundwater system. The interaction between these two systems has to be taken into account

in interpreting stream-flow following thinning of the forest (Ruprecht & Schofield, 1989). What should also be considered, if the forest itself is to be maintained in a state of ecological health, is the impact of thinning on the resident biota. Overseas, thinning has mainly been concerned with good silvicultural practice and reduction of pest attack on commercial timber (e.g., Bennett, 1968; 1971; Hedden, 1982; Nebeker *et al.*, 1985; Warriner *et al.*, 2002), rather than improving hydrology. Data for Australia are scant indeed, especially where the thinning has been carried out for hydrological purposes. Where work has been done on the impact of tree-thinning on Australian fauna, e.g., in the Box-Ironbark Ecological Thinning Trial (Arthur Rylah Institute: <http://www.dse.vic.gov.au/arthur-rylah-institute/research-themes/forest-and-woodland-ecosystems#thinning>), this has been presented as website summaries or other popular scientific formats, rather than in learned, peer-reviewed articles.

This paper seeks to address some of these deficiencies by examining the effect of hydrological tree thinning activities on the fauna of the Wungong Catchment, near Perth, Western Australia, using ants as a surrogate for other forest organisms (including plants). For some years ants have been used as biomonitoring indicators of disturbance by Australian researchers (e.g., see Majer & Nichols, 1998; Andersen *et al.*, 2002; Hoffman & Andersen 2003; Heterick *et al.*, 2013; Majer *et al.*, 2013), since they are well studied taxonomically, are diverse and numerous, are easy to collect and the different species respond in a measurable way to perturbations in their environment.

In this paper, we examine the immediate impacts of thinning on the ant fauna around the Wungong catchment in 2008 and 2009 (Figure 1), by comparing the situation prior to thinning in 2006. Further monitoring was not possible, as the thinning program was discontinued due to the combined factors of unusually dry years during the trial and an increased focus by the Western Australian Water Corporation on climate independent water sources (Water Corporation, 2011b).

METHODS

Plot descriptions

The Wungong Catchment Trial was a Water Corporation initiative to determine the ability to increase the quantity of water available for public water supply through alternative forest management practices (Loh *et al.* 2009). As part of the requirements, the Water Corporation was obliged to conduct various monitoring programmes, including fauna monitoring. Data from ALCOA's fauna monitoring programme was used as the baseline data. A similar fauna monitoring technique used by ALCOA was used in this instance in order to maintain the compatibility of fauna data collected.

Two types of catchment thinning were employed to reduce the catchment basal area to 15–18 m²/ha,

namely: (1) a non-commercial thinning (in areas where very few commercial size trees were present); and (2) a commercial logging in areas where they did exist. In both prescriptions, trees and logs to be retained were marked, and all other trees treated. Non-commercial thinning involved stem injection in the non-summer months. Commercial logging involved logging commercial-sized trees until the required BA was obtained, single tree selection, shelterwood (i.e., mature trees left standing to provide shelter for growing saplings) and the creation of gaps for regeneration. While a prescription burn after treatment was the intention for all of the treatment sites, only in Plot 5 was the burn actually carried out during the sampling period. This was done in December 2007. As at the end of 2009, the other three treatment sites had not been burnt. Some idea of the scope and impact of the fire can be gauged from Figure 2 (Plot 5).

All treated areas were situated in similar types of forest (Havel's (1975) P or P/S types). The detailed descriptions of each plot are provided in Appendix 1. Within each of the treatment types, two plots (i.e., Plots 3 and 4 – thinned non-commercially (Figure 3), and 5 and 6 – thinned by logging (Figure 4)) were set up, with two control plots (Plots 1 and 2 (Figure 5)) (marked out in non-treated forest. The six plots were established prior to treatment to provide baseline data from which change could be measured. Each plot consisted of two sets of 250 m transects, with measurements made each 20 m along the transect. An effort was made to select sites that were similar. However, logging and fire histories could not be completely matched. The thinned plots (Plots 3 and 4) had been heavily logged about 20 years previously and had not been burnt since that time. Plots 1 and 2 and Plots 5 and 6 had been selectively logged well over 50 years ago and had been prescribe burned within the past four-six years. In terms of degree of stress introduced into the environs of these four plots by the respective treatments, Plot 5 could be understood to have seen most stress as commercial logging and burning operations were both conducted in this plot within the monitoring period. Plot 6 was exposed to

Table 1. Summary of plot characteristics measured prior to treatment in 2006 and post-treatment in 2009 (in brackets) (adapted from Kabay, 2009).

	Control		Thinned		Thinned/logged and burned	Thinned and logged
2006(2009)	Plot 1	Plot 2	Plot 3	Plot 4	Plot 5	Plot 6
BA (Basal Area (m ² /ha))	41.5(46)	31*(34)	27.5(11.5)	29(13)	37.5(17)	31(19.5)
% Canopy cover	31.5(36)	49.5(48)	32.5(16.5)	30(15)	46.5(20)	49(21)
SPH (Stems per ha)	1,980(1500)	510(500)	1,886(161)	781(155)	702(317)	878(275)
% Total Forest Cover	131.1(104.8)	138(103.2)	46.3(37.9)	63.1(52.1)	75.2(52.9)	88.7(66.8)
% cover in 0–25 cm strata	16.4(9.6)	14.6(8.6)	3(2.8)	5.2(4.2)	8.5(3.4)	11.8(7.8)
% cover in 25–50 cm strata	23.1(13.8)	17(9.8)	3.1(3.4)	4(4.4)	11.8(5.1)	10.2(6.7)
% cover in 50–100 cm strata	24.4(17.3)	18.6(12.9)	3.8(4.2)	9.8(8)	11.2(9.3)	11.1(8.9)
% cover in 100–600 cm strata	21.8(18.7)	32.9(23.9)	19.2(12.5)	20.5(15.3)	23.9(15.3)	21.8(15.2)
% cover in >600 cm strata	45.4(45.4)	54.9(48)	17.2(15)	24.4(20.2)	19.8(19.8)	33.8(28.2)
Logs<10 cm diam	3.8(2.8)	4.5(3.4)	0.6(1)	1.6(1.6)	2.5(1.7)	8(9.1)
Logs10–30 cm diam	4.3(3.3)	4.5(3.3)	0.3(1.3)	2.6(2.7)	2.9(3.1)	6(9)
Logs>30 cm diam	7.1(3.9)	8.3(4.7)	0.7(1)	4.3(3.5)	3.2(3.9)	5.4(6.9)
Litter%	28.4(70.8)	26.2(67.3)	38.7(64.9)	47.5(64.3)	29.1(40.4)	67.9(63)

* mean of four measurements done early 2006

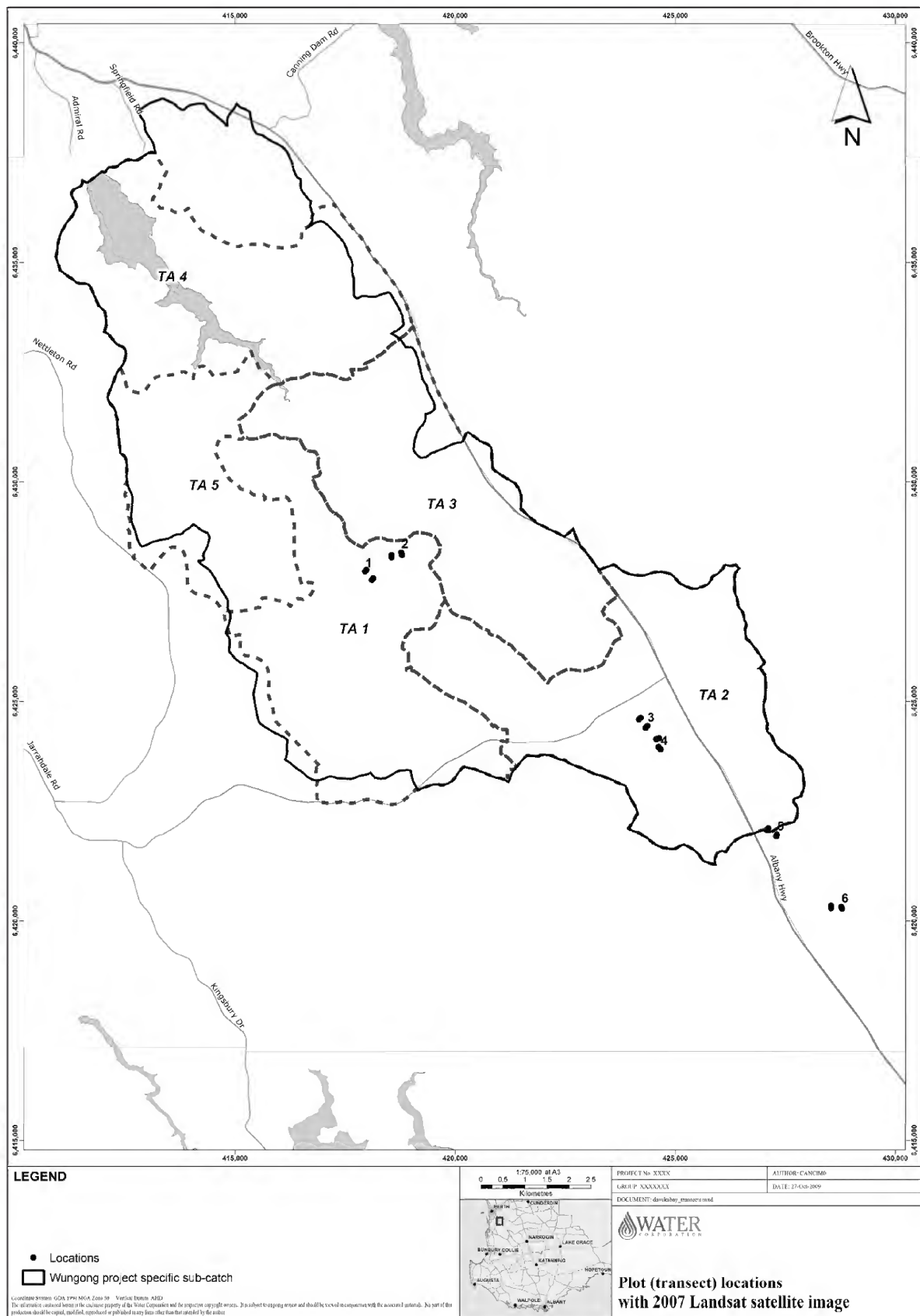


Figure 1. Location of the Wungong Catchment tree thinning project.

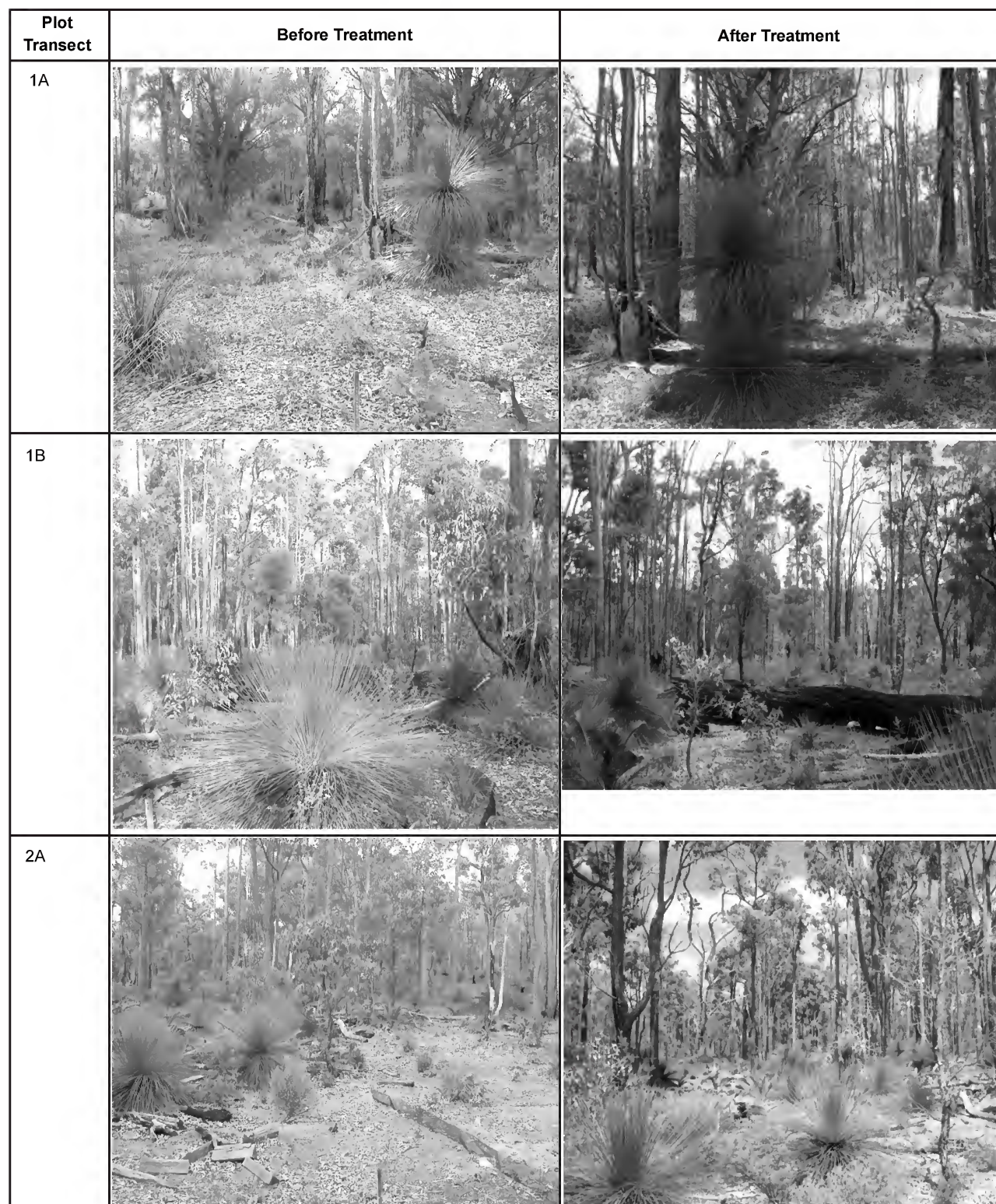


Figure 2. Photographs taken in 2009 of two forest Controls (Plots 1 and 2) two thinned plots (Plots 3 and 4), the thinned, logged and burned plot (Plot 5) and the thinned and logged plot (Plot 6). View A is the commencement of the transect (looking up the transect) and view B is the end of the transect (looking down the transect).

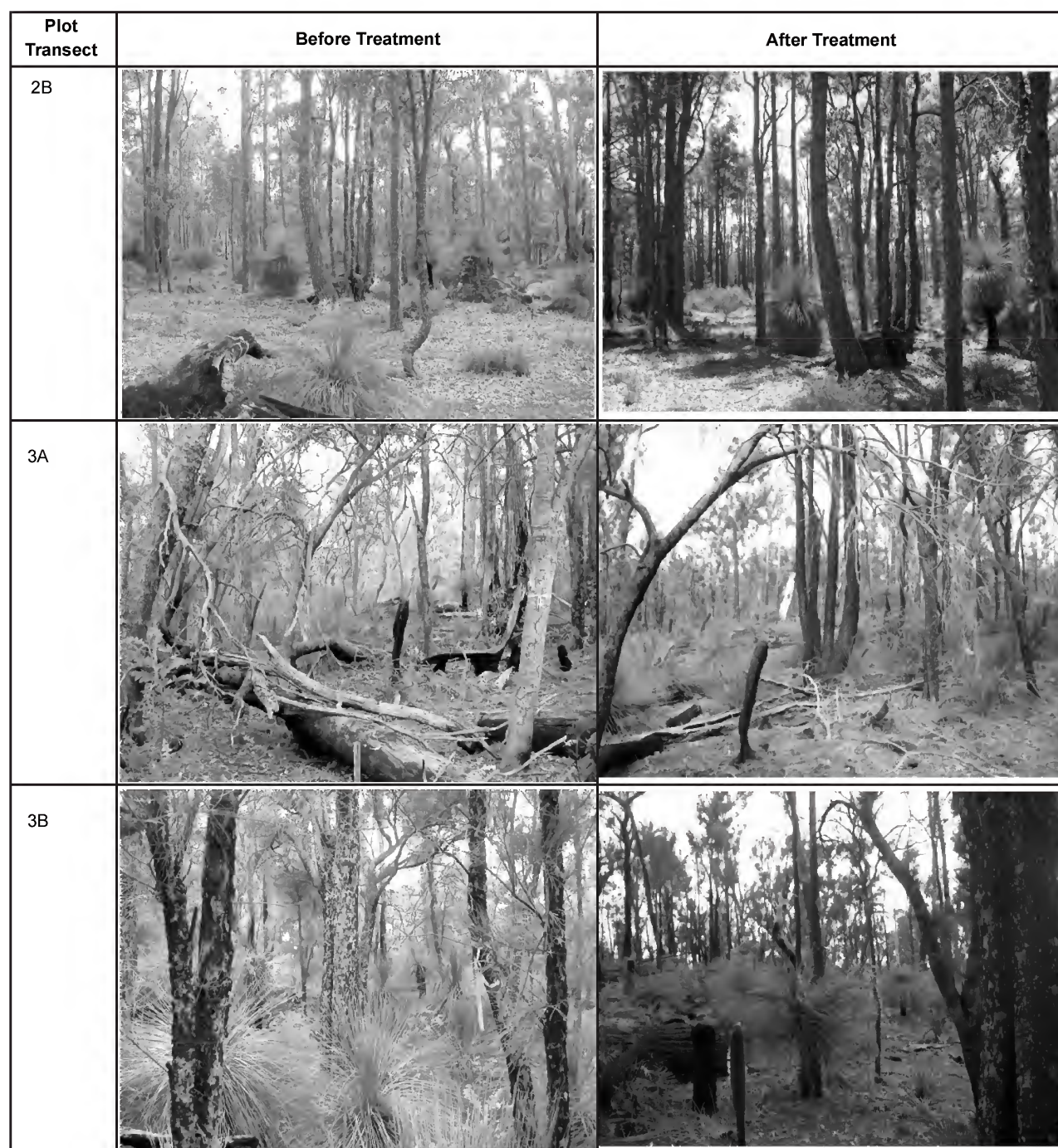


Figure 2 (cont.) See p. 104 for caption details.

commercial logging, and hence greater environmental stress than Plots 3 and 4, which were exposed to non-commercial thinning only. The effect on fauna was therefore hypothesised to form a descending gradient from Plot 5 to Plot 6, through to Plots 3 and 4 and down to the control Plots 1 and 2.

Prior to stem injection and logging operations, basic vegetation characteristics relevant to the study were measured (a summary is given in Table 1). (The low values for some parameters in Plots 3 and 4 reflect a heavy logging operation that was carried out on these sites during the time that adjoining areas were being

mined for bauxite. This had limited the basal area that might have been expected given the many years that had passed since the last burn.)

Invertebrate sampling

While other fauna groups were also sampled, only ants are reported on here. Ants were collected in 20 pitfall traps (42 mm diameter x 100 mm depth) set in each plot, i.e., 120 traps in all. All traps contained 50 ml of Galt's Solution to preserve specimens. All ant sampling was performed in summer, when ant activity is greatest. In 2006, traps were opened between 7th and 8th of

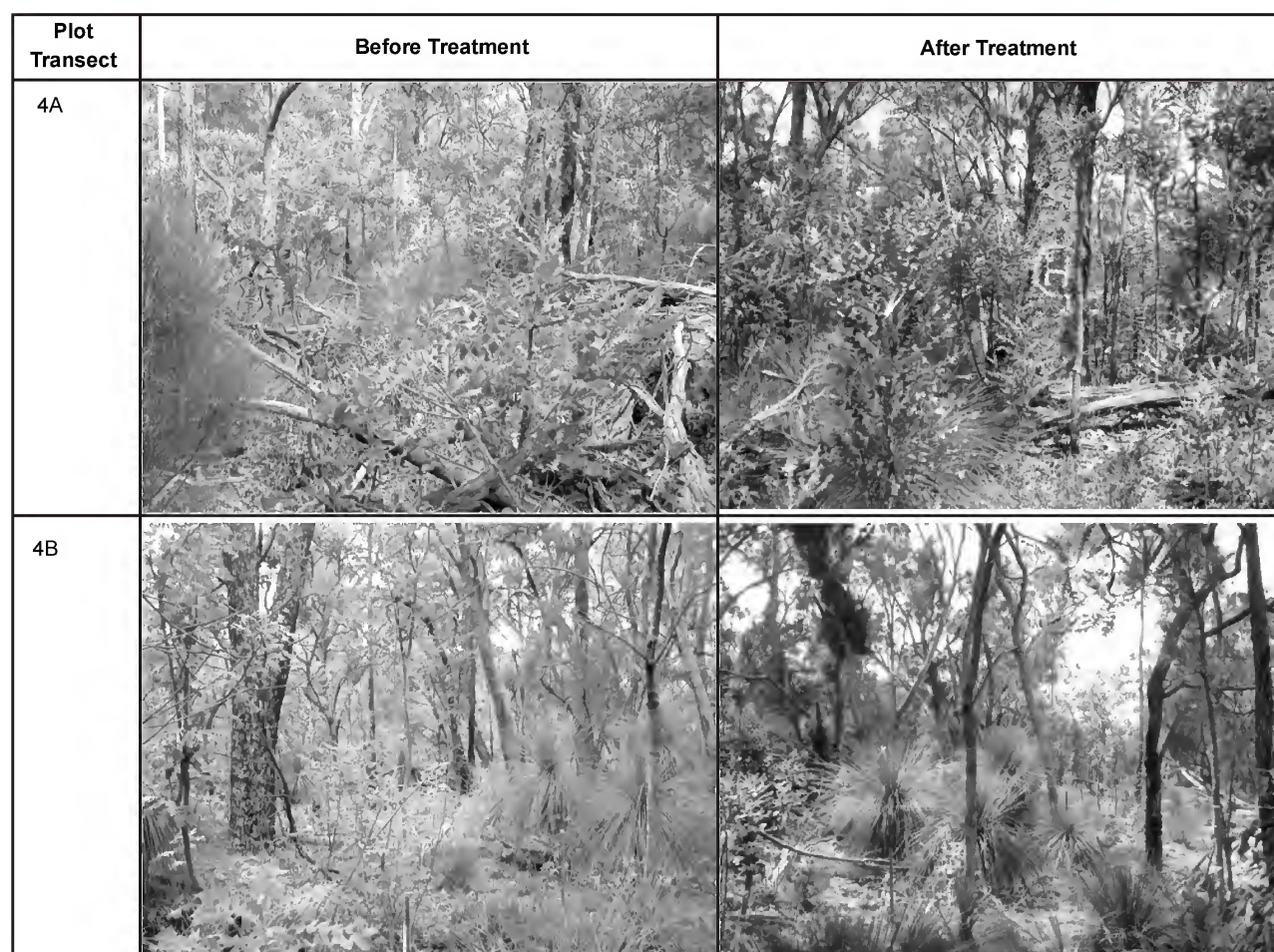


Figure 2 (cont.) See p. 104 for caption details.

March and taken up on 14th or 15th March. Two post-treatment monitoring operations were performed for the same sampling length in 2008 and 2009, respectively. However, no monitoring was conducted in Plots 2 and 4 during 2008. The precise date of setting up of the traps and collection of the catch is not known for 2008, which took place over one week in January of that year, but the 2009 catch was collected from traps set up on 7th or 8th of March 2009 and taken up on 14th or 15th March 2009, as had been done in 2006. Relevant rainfall and temperature records for the known dates of collection (7th – 15th March 2006 and 2009) can be found in Appendix 2. Specimens were identified by the first author at Curtin University and voucher specimens for the species found are deposited in the Western Australian Museum.

Data analysis

The numbers of ants within each species were tabulated for each plot and summarised as ant abundance, ant species richness, species diversity (Shannon-Wiener index) and evenness (calculated by dividing the Shannon-Wiener index by the square root of species richness). These summaries were then presented as histograms for each plot over the three sampling periods. Regrettably, the absence of 2008 data for two of the plots means no comparative quantitative statistical analysis such

as comparison of mean values is possible. Instead, a multivariate analysis of the sites was performed using the package PAST 3.0 (Hammer, 2013). Using this package, a non-metric multidimensional scaling (NMDS) ordination, based on Bray-Curtis matrices data, was performed. In order to represent the trajectories in each plot over time, the corresponding points have been joined up and the similarity index values drawn in. This is necessary because the length of a line on a two-dimensional diagram may not accurately represent the similarity in multi-dimensional space. It should be remembered that 2008 data do not exist for Control Plot 2 or Thinned Plot 4. However, the degree of similarity between the 2009 and the pre-treatment 2006 data is of the greatest interest, along with the magnitude of the change in 2008 after the forest treatment, as these represent longer-term and immediate responses, respectively.

RESULTS

A total of 6681 ants belonging to 88 species, 31 genera and eight subfamilies was collected over the three seasons (Table 2). Forty-six species were trapped in Plot 1, 24 in Plot 2, 51 in Plot 3, 43 in Plot 4, 42 in Plot 5 and 38 in Plot 6. The species data for Plots 2 and 4, however, should be treated with caution because no trapping was undertaken on these sites in 2008.

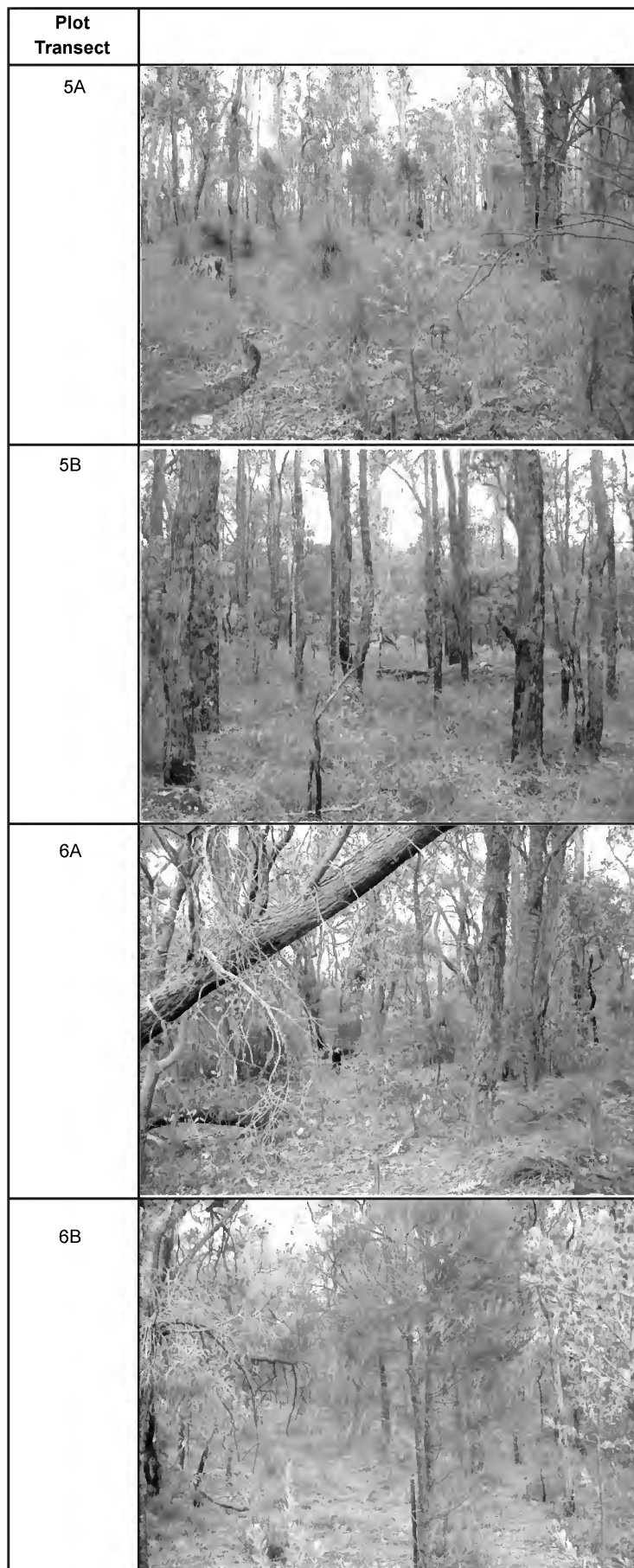


Figure 2 (cont.) See p. 104 for caption details.

Table 2. Ant species collected in the six forest plots over a three year sampling period (2006, 2008 and 2009).

(Year)	Control 1			Control 2			Thinned 3			Thinned 4			Thinned/logged and burned 5			Thinned and logged 6		
	2006	2008	2009	2006	2008	2009	2006	2008	2009	2006	2008	2009	2006	2008	2009	2006	2008	2009
Cerapachyinae																		
<i>Cerapachys flammeus</i> (Clark)	1				*						*							
<i>Cerapachys gilesi</i> (Clark)					*	17					*	1						
<i>Cerapachys</i> sp. JDM 1040					*		12		2		*							
Dolichoderinae																		
<i>Anonychomyrma fornicata</i> (Emery)		1	1		*	1				1	*	2		7	1			1
<i>Anonychomyrma itineransperthensis</i> Forel					*						*		1			16	1	2
<i>Anonychomyrma nitidiceps</i> André	1		2		*	6		5	1		*	2		1			6	13
<i>Arnoldius flavus</i> (Crawley)					*	1					*							
<i>Doleromyrma darwiniana</i> (Forel)	52	43	44		*		68	6	5	27	*	4	22	3	6	7	2	4
<i>Doleromyrma rotnestensis</i> (Wheeler)					*				4		*	38			1			
<i>Dolichoderus ypsilon</i> Forel					*						*		1	1				
<i>Iridomyrmex bicknelli</i> Emery					*		10		37	103	*	73			284			
<i>Iridomyrmex calvus</i> Emery	1				*					1	*		1					
<i>Iridomyrmex conifer</i> Forel					*		48	33	280	5	*				13			
<i>Iridomyrmex discors</i> Forel					*		4				*							
<i>Iridomyrmex innocens</i> Forel					*				1	1	*							
<i>Iridomyrmex mjobergi</i> Forel					*				7		*							
<i>Iridomyrmex omalonotus</i> Heterick & Shattuck					*			5			*							
<i>Iridomyrmex splendens</i> Clark		2	1		*			11	8		*	9			3			
<i>Ochetellus glaber</i> gp. sp. JDM 19			3		*		1	2		1	*	2		1	1	1	2	
<i>Tapinomaminutum broomense</i> Forel	3	11			*	1				3	*	1	3			12	4	1
Ectatomminae																		
<i>Rhytidoponera inornata</i> Crawley	19	4	27		*		1	1		8	*	16		1	8			1
<i>Rhytidoponera metallica</i> (F. Smith)	36	35	104		*	2	48	84	38	62	*	36	2		25		3	1
<i>Rhytidoponera punctigera</i> Crawley					*			8			*							
<i>Rhytidoponera rufonigra</i> Clark					*		2		4	1	*							
<i>Rhytidoponera violacea</i> (Forel)	10	25	9	1	*	2	7	2	16	29	*	45			1			
Formicinae																		
<i>Camponotus claripes</i> Mayr		1		1	*			1			*						2	
<i>Camponotus marzens</i> Forel	2				*		3	1	1	4	*	2						
<i>Camponotus michaelsoni</i> Forel			3	2	*	2	7		2		*				1			
<i>Camponotus minimus</i> Crawley		4			*	2	1		2	1	*					1		10
<i>Camponotus rudis</i> McArthur	1				*						*							
<i>Camponotus prostans</i> Forel					*	1	4		3	1	*	1		5	3			
<i>Camponotus scotti</i> McArthur					*				1		*							
<i>Camponotus terebrans</i> (Lowne)			1		*			1			*		3	12	43			
<i>Melophorus insularis</i> Wheeler	29	25	18	3	*	5	21	46	36	43	*	50		6	17	3	8	38
<i>Melophorus ludius</i> Forel	18	9	30		*		37	9	71	41	*	192	1	11	11		2	11
<i>Melophorus turneri perthensis</i> Wheeler					*		6	10	11	23	*	62						
<i>Melophorus</i> sp. JDM 898					*						*	1						
<i>Notoncus cf capitatus</i> Forel					*					2	*	1						
<i>Plagiolepis squamulosa</i> Wheeler				3	*						*			1			2	
<i>Plagiolepis</i> sp. JDM 189					*						*		1				1	
<i>Prolasius antennatus</i> McAreavey	2				*						*							
<i>Prolasius reticulatus</i> McAreavey		1			*					1	*							

<i>Stigmacros aemula</i> Forel					*					5	*	1					2	
<i>Stigmacros brachytera</i> McAreavey					*			1			*							
<i>Stigmacros clarki</i> McAreavey					*		1				*		1					
<i>Stigmacros debilis</i> (Bolton)	1				*						*							
<i>Stigmacros epinotalis</i> McAreavey	1				*						*		1			5		2
<i>Stigmacros glauerti</i> McAreavey					*					2	*	1	3		5	5		8
<i>Stigmacros occidentalis</i> (Crawley)					*					1	*		1				1	
<i>Stigmacros</i> sp. JDM 115		2	22	21	*	23	4	1			*		7	1	1	5		9
<i>Stigmacros</i> sp. JDM 443		18			*						*			12				
<i>Heteroponera imbellis</i> (Emery)					*						*					1	1	
Myrmeciinae																		
<i>Myrmecia callima</i> (Clark)					*			2			*							
<i>Myrmecia chasei</i> Forel					*				1		*	1						
<i>Myrmecia mandibularis</i> F. Smith	1				*				1		*							
<i>Myrmecia vindex</i> F. Smith	1				*						*							
Myrmicinae																		
<i>Adlerzia froggatti</i> (Forel)			1		*						*							
<i>Anisopheidole antipodum</i> (F. Smith)	5	1			*		2		5		*	3						1
<i>Austromorium flavigaster</i> (Clark)					*						*		2					2
<i>Colobostruma mellea</i> Shattuck	1		38		*						*							
<i>Crematogaster dispar</i> Forel	1		1	4	*	9					*		10	15		18	4	5
<i>Crematogaster laeviceps chasei</i> Forel					*		1				*				1			
<i>Crematogaster queenslandica</i> gp. sp. JDM 428			1		*						*					2	90	
<i>Meranoplus ferrugineus</i> Crawley	1				*		2			1	*							6
<i>Meranoplus rugosus</i> Crawley	9				*		4		1	1	*	4						
<i>Meranoplus</i> sp. JDM 74					*		5		1	1	*	7						
<i>Meranoplus</i> sp. JDM 491					*						*		1	1		1		
<i>Meranoplus</i> sp. JDM 677					*						*							5
<i>Meranoplus</i> sp. JDM 1107					*			3			*							1
<i>Monomorium fieldi</i> Forel					*						*			1				2
<i>Monomorium hildebrandti</i> gp. sp. JDM 438	2				*					2	*							
<i>Monomorium laeve</i> Mayr		21			*	1		13		16	*	2		4	7		5	
<i>Monomorium laeae</i> Forel	6	4	9	17	*	8	2		3	3	*		9		6		5	
<i>Monomorium sordidum</i> Forel	106	66	99	224	*	96	43	1	42	16	*	183	22	19	116	132	43	265
<i>Monomorium sublamellatum</i> Heterick					*	1					*							
<i>Monomorium sydneyense</i> Forel	61	9	8	8	*	16	312	11		46	*		11	10	9		6	
<i>Monomorium sydneyense</i> complex sp. JDM 101			62		*	13			124		*	161		2	60			
<i>Pheidole ampla perthensis</i> Crawley	5	14	3	10	*	13		7			*		1	12		29	45	54
<i>Pheidole rugosula</i> Forel	2		36		*						*							
<i>Pheidole</i> sp. JDM 164					*		4		2	45	*	24			4		1	
<i>Solenopsis clarki</i> Crawley	9		2		*	1	3		2	5	*	4	1			1	1	
<i>Tetramorium impressum</i> (Viehmeyer)		1			*					5	*	3						
<i>Tetramorium striolatum</i> Viehmeyer	2				*		9		8	11	*	19			2			
Ponerinae																		
<i>Austroponera rufonigra</i> (Clark)					*		1				*							
<i>Brachyponeralutea</i> (Mayr)			1		*		1	3	5	2	*	3				1		1
<i>Hypoponera congrua</i> (Wheeler)	1				*						*							
<i>Leptogenys neutralis</i> Forel				1	*						*						1	
<i>Platythyrea micans</i> (Clark)					*		1	1	1		*							
Total abundance	390	293	530	295	*	221	675	266	728	527	*	954	105	126	629	246	246	450

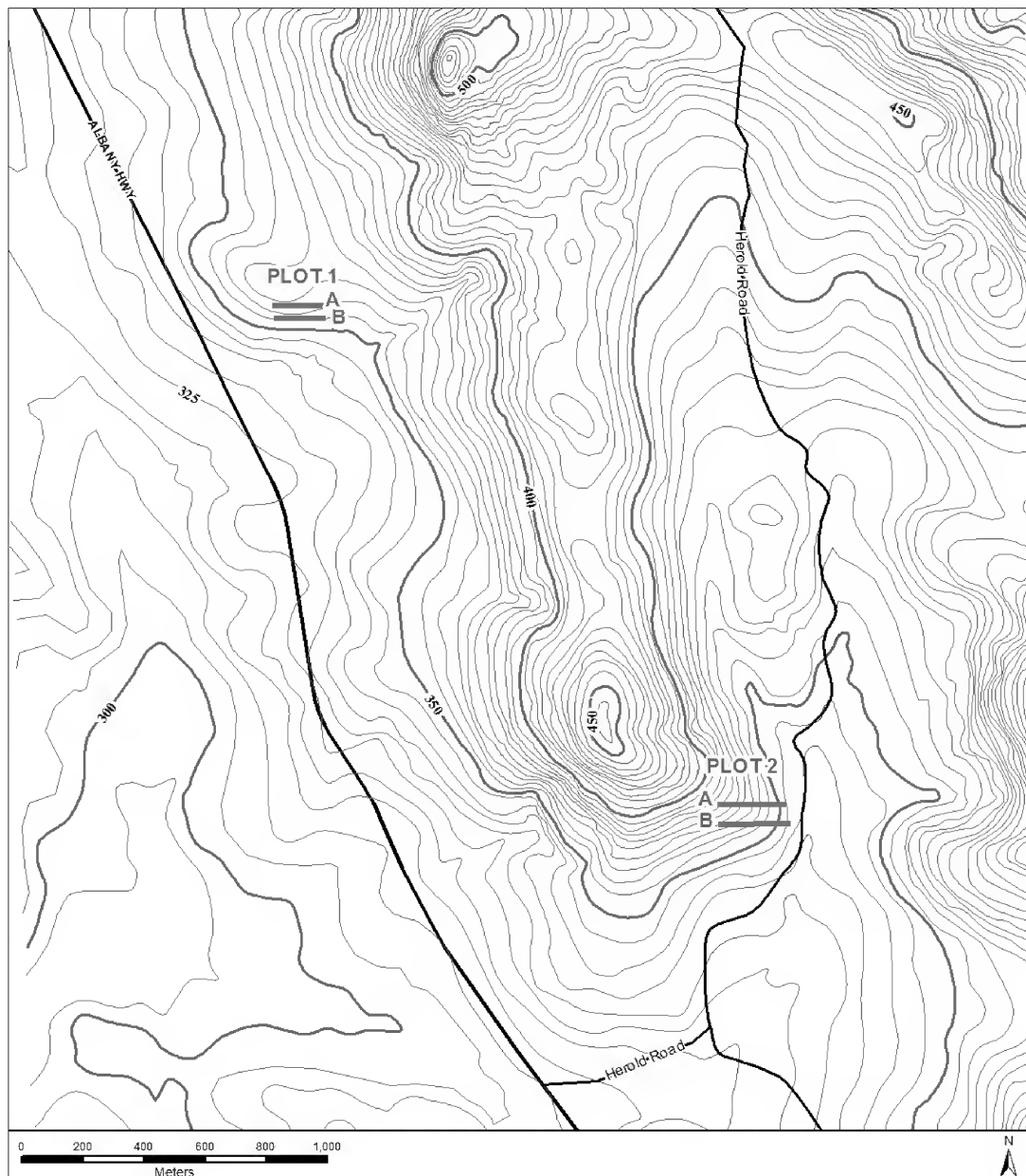


Figure 3. Catchment thinning treatments plots 3 and 4 (adapted from Kabay, 2006, Figure 2).

Ant summaries

Histograms of ant abundance, species richness, and evenness over the three sampling years are shown in Figures 5, 6 and 7, respectively; species diversity is not presented here as it is derived from components of richness and evenness and provides no additional information to that shown in the other graphs.

Ant abundance varied from 954 (Plot 4 in 2009) to as little as 105 (Plot 5 in 2006) (Figure 6). Ant abundance increased from the first survey to the last in one of the control plots (Plot 1), but not in the other. However, abundance increased from pre-treatment to the 2009 survey period in all four thinned or logged plots, with the increase being most pronounced in the thinned, logged and burned plot (Plot 5).

The greatest richness was found in Plot 4 in 2006,

when 37 ant species were recorded, and the lowest richness was discovered in Plot 2 (also in 2006) when only 12 ant species were recorded. A simple regression of the abundance versus the richness data using the PAST program mentioned above indicated high abundance was associated with high diversity ($r^2 = 0.458$, $r = 0.676$, $p = 0.004$; Ordinary Least Squares algorithm). Despite this correlation, the trends in the treated plots did not reflect the trends in abundance so markedly, with only slight increases in richness by the time of the last survey in thinned Plot 3 and thinned, logged and burned Plot 5 and thinned and logged Plot 6; there was a slight decline in richness in thinned Plot 4.

A very low evenness index of only 0.41 was recorded for Plot 2 in 2006 while, at the other end of the scale, Plot 5 saw a high evenness index of 0.86 in 2008 (Figure 7).

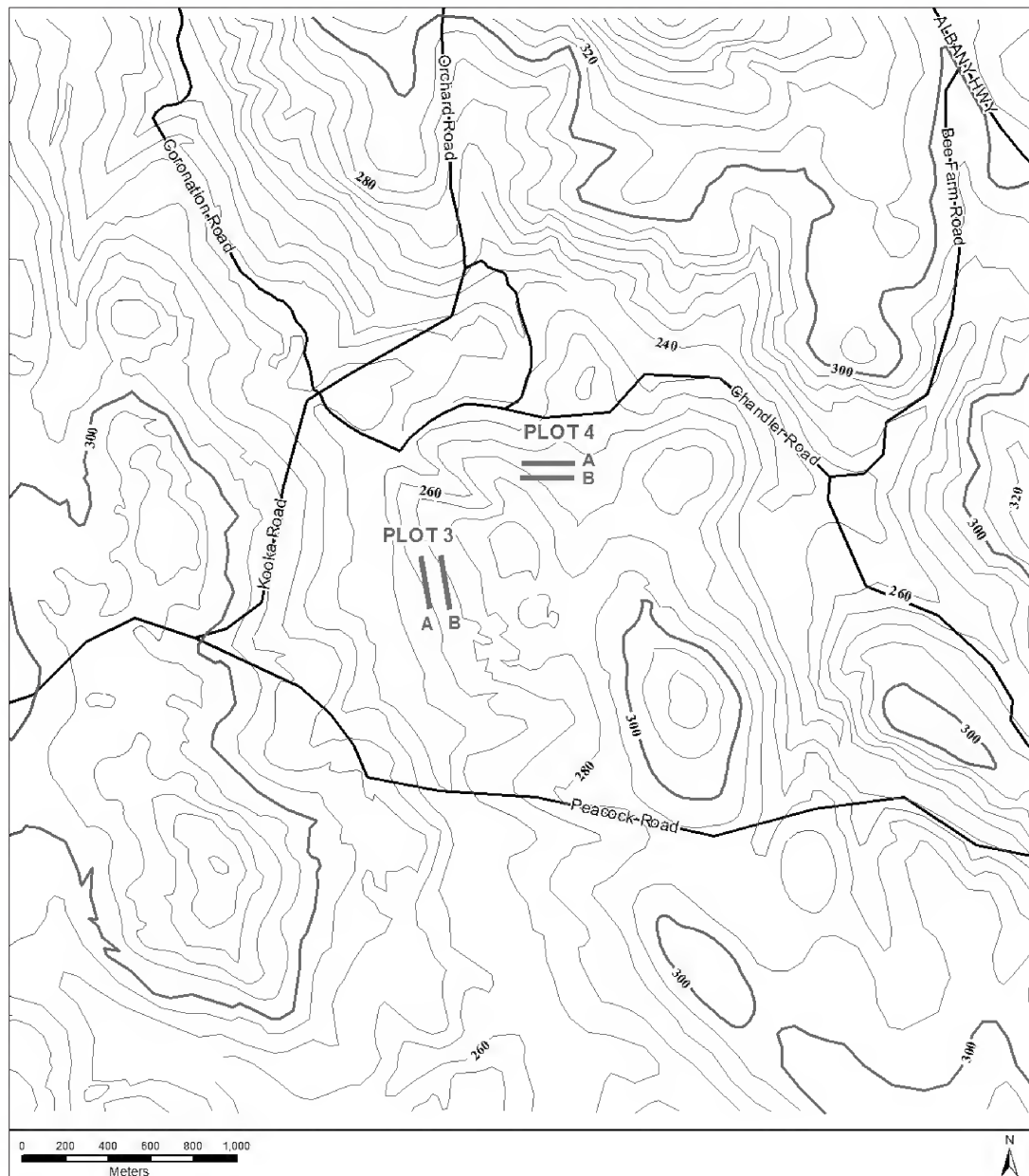


Figure 4. Catchment thinning, logging and burning treatment (Plot 5) and thinning and logging treatment (Plot 6) (adapted from Kabay, 2006, Figure 3).

Evenness was higher in the 2009 surveys than in 2006 for both control plots. However, it declined in the two plots that had been logged (Plots 5 and 6), stayed the same in thinned Plot 3 and increased in thinned Plot 4.

Identification of changes in the presence of individual species is complicated by the fact that trapping positions were not identical in each survey. Furthermore, even when placed in an identical position, the actual species caught are subject to chance foraging events, meaning that even if there are no changes in species present, the same set of species may not necessarily be sampled. Nevertheless, an analysis of the species lost or gained in 2009, three years after treatment, may provide some indication of the degree of species turnover in control and treated plots.

The two control plots (Plots 1 and 2) incurred two and three losses, and six and 11 gains, respectively. These losses and the relatively high number of gains illustrate the dynamic nature of ant assemblages. The losses in treated plots 4–6 were three, 11, three and three in that order, and the gains were six, six, four and six. Thus, apart from the high number of gains in thinned plot 4, these losses and gains were within the range found in the control plots.

Considering the changes in functional group (*sensu* Greenslade and Thompson, 1981, Andersen, 1990) is informative, however. If profiles for combined control plots are compared with combined treatment plots, there is evidence of a higher degree of gain in the treated plots of species from the Dominant Dolichoderinae (e.g.,

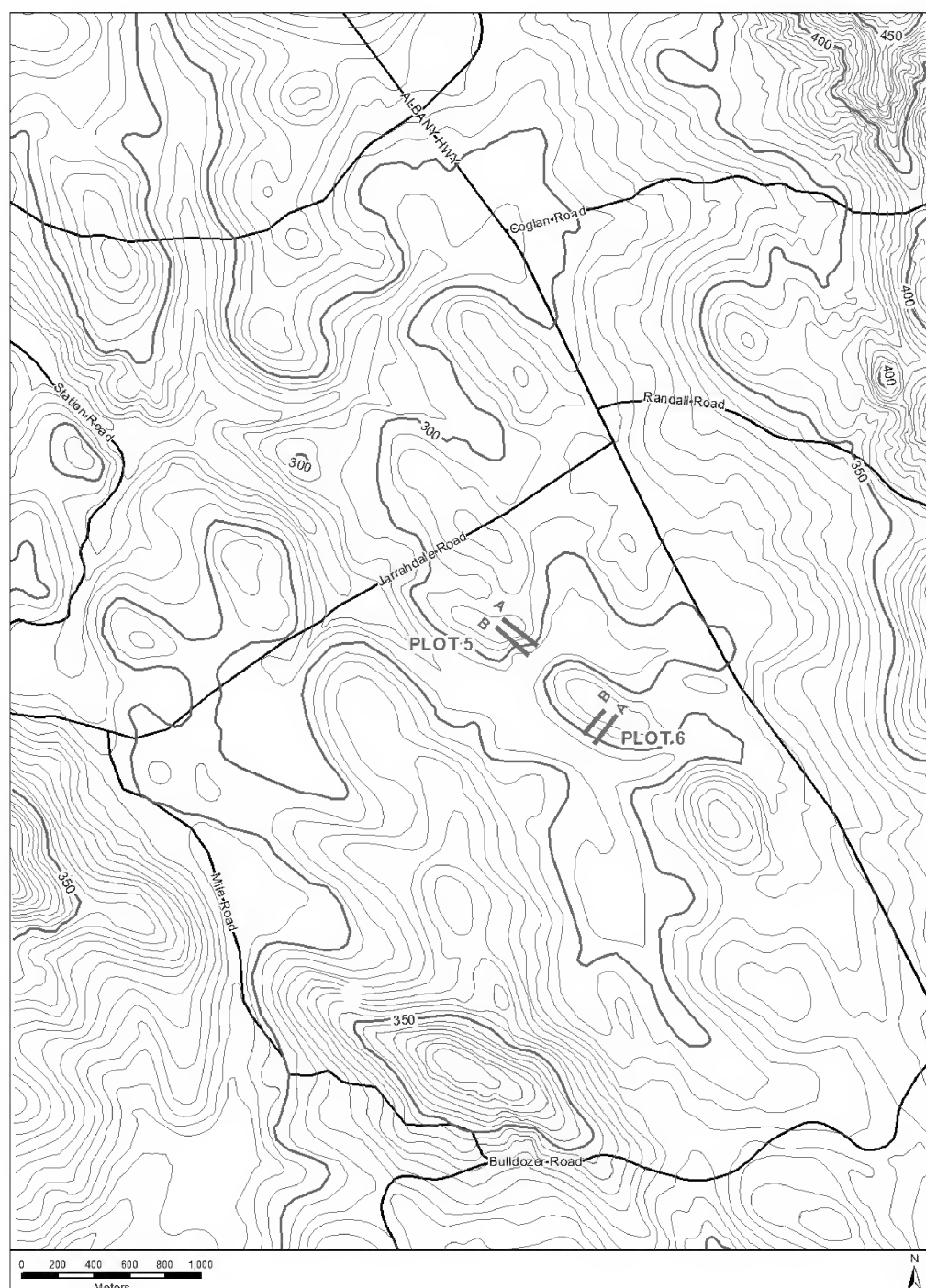


Figure 5. Control Plots 1 and 2 (adapted from Kabay, 2006, Figure 4).

Iridomyrmex spp.), Hot Climate Specialist (e.g., *Melophorus* spp.) and Opportunist (e.g., *Rhytidoponera* spp.) groups when compared with the situation in the controls.

Multivariate analysis

The four plots sampled in 2008 all exhibited shifts in ant assemblage composition from that present in 2006, with similarity being lowest in the thinned plus logged plots (0.642 and 0.669 in Plots 5 and 6, respectively), intermediate in thinned Plot 3 (0.690) and highest in

control plot 1 (0.840). Over the longer term, both control plots exhibited the highest similarity (0.802 and 0.881 in Plots 1 and 2, respectively), intermediate values in thinned Plots 3 and 4 (0.746 and 0.776, respectively) and lowest similarity in the thinned plus logged plots (0.663 and 0.708 in Plots 5 and 6, respectively). Of particular note is the fact that the logged and thinned plot that had also been burnt (Plot 5) had the lowest similarity value. Thus, overall, increasing disturbance seems to result in the greatest change in ant assemblage composition (Figure 9).

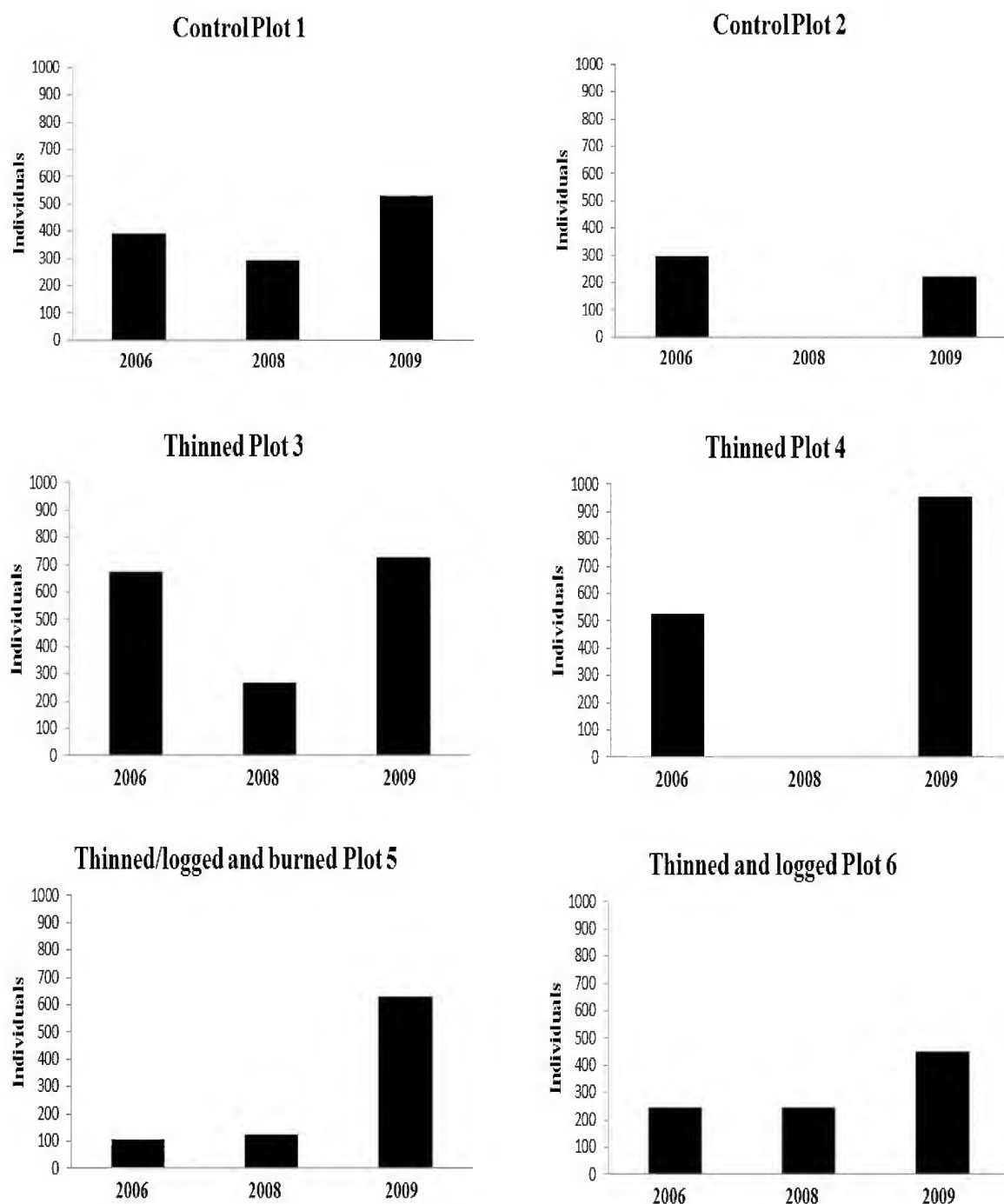


Figure 6. Ant abundance for the six plots.

DISCUSSION

General Trends

Any interpretation of the data is limited because of the incomplete sampling, and also the different months in which the ants were collected (January 2008 and March 2006 and 2009). While thermophiles are normally most active in mid- to late-summer when ground surface temperatures are at their highest, other species may be winter-active or forage early in the morning or at dusk when ground surface temperatures are

cooler. Regrettably, this also means there is no exact correspondence between the datasets for the three years. Some idea of the surface conditions can be obtained from a composite picture of climate records from Karnet (Bureau of Meteorology 2014) and the Wungong Catchment (incomplete) (Kabay 2006, 2009). These indicate that rainfall in January 2008 was 0.0 mm (Kabay, 2009) and rainfall for March 2006 was 7.6mm (all but 0.2 mm falling before 15th March) (Kabay, 2006) and for March 2009 it was 8.4 mm (all falling after the 15th March) (Kabay, 2009). Average temperature maxima

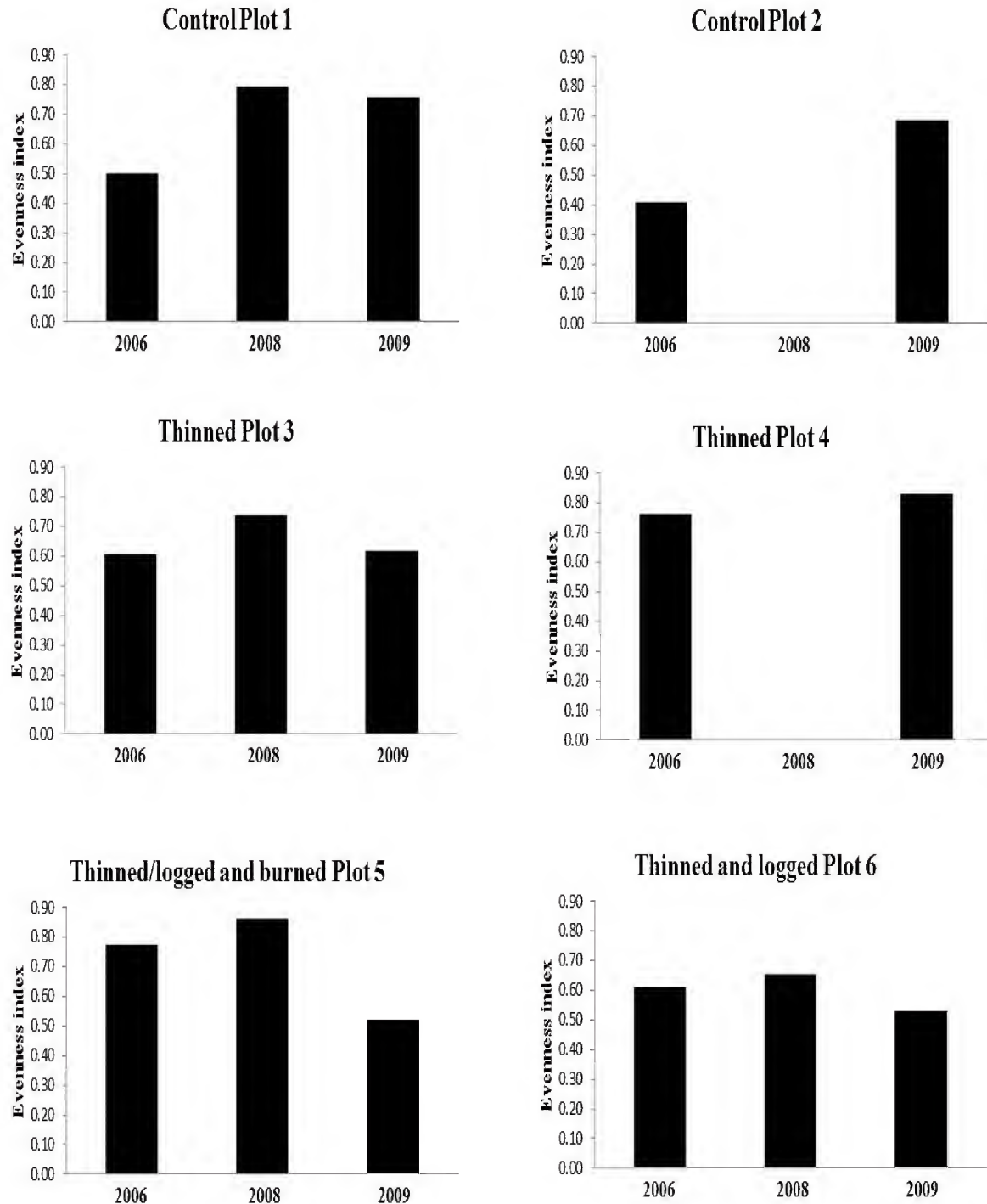


Figure 7. Shannon ant evenness index for the six plots.

for the same periods were 28.5°C (March 2006) (Kabay, 2006), 31.4°C (January 2008) and 26.9°C (March 2009) (Bureau of Meteorology). Average temperature minima were 14.3°C (March 2006) (Kabay, 2006), 15.9°C (January 2008) and 13.1°C (March 2009) (Bureau of Meteorology 2014). (Since Karnet is only 11 km away from the plots, temperature records from the two localities are likely to be comparable).

What these differences mean in effect is that species sampled in January (i.e., in 2008) may not be identical

with those that would have been taken had the sampling been conducted two months later. While we make the tentative suggestion that the almost uniformly higher abundance recorded in the March sampling of 2009 may be related to slightly cooler, moister conditions, there are the additional caveats mentioned under the Results; i.e., the inherently stochastic nature of foraging activity in ants and the fact that the exact position of the pitfall traps may not have been identical in all years. However, the large turnover in ant species within the control plots suggests real differences in the ant fauna over the

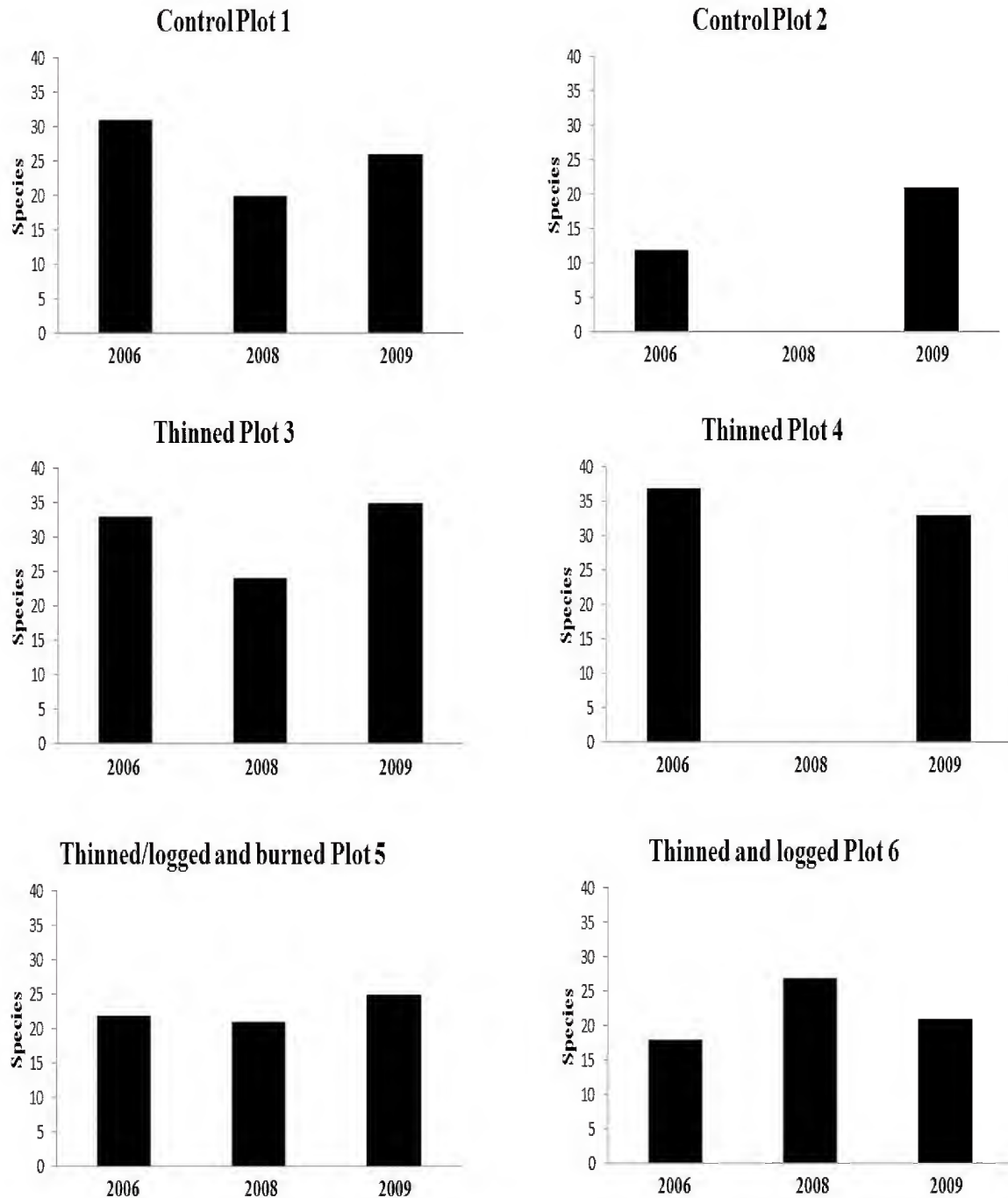


Figure 8. Ant species richness for the six plots.

three years and not just factitious dissimilarities due to the variables mentioned. What we also suggest is that although ant abundance data are amenable to being skewed by the month of collection (with ant activity being at a peak in late spring and declining by late autumn) the species richness is less so, since the colonies of most species will be dispersed over a relatively large area of suitable terrain and soils.

Keeping the various provisos in mind, the lower number of ant species collected in Plots 1, 3 and 5 may be a function of the dry, hot conditions obtaining during January 2008. More species appear to have been active in

March 2006 and March 2009. Individual taxa and suites of taxa are discussed below, but clear trends are mostly lacking in the treatment plots compared with the control plots, although such groups as specialist predators were generally absent from the treatment plots. What can be detected is an upsurge in species richness in Plot 6 (discussed at greater length below) in 2008, followed by a reversion to near former levels in 2009. This is in marked contrast to the catch in 2008 in Plots 1, 3 and 5. The characteristics of the types of ant species collected in Plot 6 in 2008 (mainly small myrmecines) appears to be independent of time of year, rainfall or pitfall trap

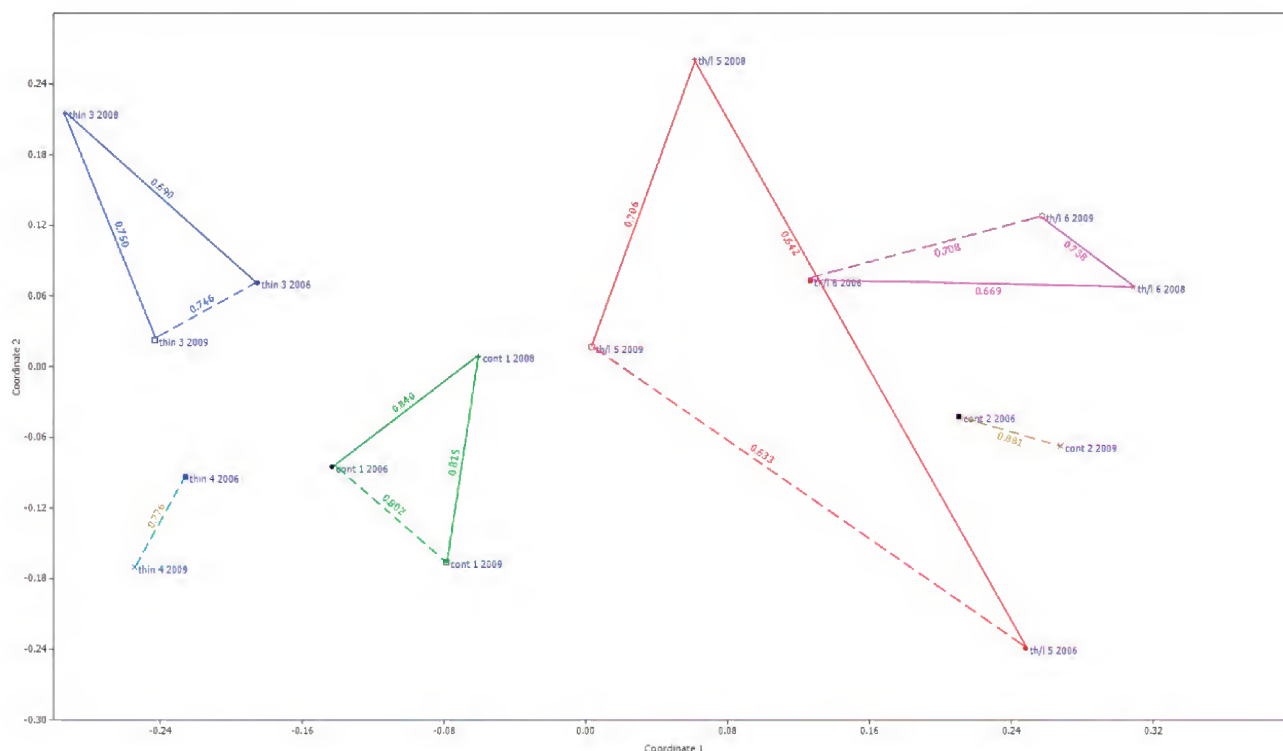


Figure 9. Ordination by non-metric multidimensional scaling (NMDS) of log-transformed ant data from six woodland plots; two controls (brown and green lines, respectively), two thinned (blue and aqua, respectively), one thinned and logged (red) and one thinned, logged and burned (pink). Direction of vectors: 2006→2008→2009. Proportions placed next to the vectors represent Bray-Curtis similarity indices (lower figures reveal increasing dissimilarity; higher figures increased similarity).

location. These species may have been temporarily favoured by extraction of commercial logs without the burning conducted in Plot 5, but this advantage did not extend into the following year, by which time some vegetative regrowth had occurred. Plot 5 also reveals interesting data (discussed below), but overall species richness was not affected. Within the control plots some gradual recovery of the respective ant assemblages might be expected following the burn several years before, but given the complicating factors connected with the way the project was conducted, it is hard to identify specifics from the data where that recovery may have occurred.

In all, general observations in this case are hampered due to deficiencies in the design of the project, incomplete sampling, the lack of temporal replication and, to a lesser extent, various other factors (i.e., different months in which sampling was conducted, questionable location of the pitfall traps and the inherent stochasticity of ant foraging behaviour). However, more substantial comments can be made when the catch is analysed at the functional group and species level.

Species Trends

Species variation in the plots over the three years that comprised the sampling period was not confined to the treated plots. The species turnover in control Plot 1 is significant; especially since three solitary predators (*Cerapachys flammeus* (Clark), *Myrmecia mandibularis* F. Smith and *Myrmecia vindex* F. Smith) and a cryptic predator (*Hypoponera congrua* (Wheeler)) were not

recovered in 2008 or 2009. Along with these species 10 other ants, several of them rare and localised in their distributions, were not collected in 2008 or 2009. Of those ants that were novelties in 2008 or 2009 (16 spp. in all), there were no Hot climate specialists or Specialist predators and *Camponotus* species and small, generalised or seed-gathering myrmecines predominated (eight species). This suggests that natural environmental phenomena may have played a role in species persistence within the three year period in this particular plot. In general, ground cover and number of logs declined and total forest cover declined significantly in this plot. Possibly, this reduced herbivorous and sap-sucking insect populations and so the potential forage for large ant species that require larger prey and greater quantities of nectar and honeydew to provision their colonies. However, litter between 2006 and 2009 was greatly increased (from a percentage cover of 28.4 to 70.8). While this may have advantaged smaller, shade or litter-loving species such as the two *Stigmacros* species, *Adlerzia froggatti* (Forel) and *Prolasius reticulatus* McAraeve that were collected in 2008 and 2009, the data are ambiguous, since two other *Camponotus*, two *Stigmacros* and a *Prolasius* species collected in 2006 were not seen subsequently. The presence of the Opportunists *Tetramorium impressum* (Viehmeyer) and *Ochetellus glaber* sp. JDM 19 (to which should probably be added *Camponotus terebrans* (Lowne)) in 2008 and/or 2009 is an indication that some enhanced degree of disturbance – perhaps caused by opening up of the canopy – was also a feature of this period. Again, some caution is required in

interpreting the data as four Opportunists (to which the myrmicine *Monomorium sordidum* Forel should probably be added) and two thermophilic *Melophorus* were among the handful of ant species collected in all three sampling events. The large number of Opportunists (at least seven in all) and thermophilic species (five) collected from this plot is a likely indicator that the ant fauna it contained is not indicative of a pristine environment, and the fairly recent burn history was still playing out when the sampling was done.

Plot 2 was characterised by lower ant richness in 2006 (just 12 species). No sampling was done in 2008 but the ant richness had climbed to 21 in 2009. While one Cryptic species (*Plagiolepis squamulosa* Wheeler) and one Specialist predator (*Leptogenys neutralis* Forel) were not seen after 2006, no fewer than 12 new species were recorded in 2009. These included several myrmicines and dolichoderines, the Specialist predator *Cerapachys gilesi* (Clark) and the very rare and possibly fossorial myrmicine *Monomorium sublamellatum* Heterick. Vegetation values showed an overall decline compared with Plot 1; litter increase was less (26.2–67.3% compared with 28.4% – 70%) and % cover was reduced at the 0–25 cm strata level (14.6–8.6% compared with 16.4% – 9.6%) and the 100–600 cm strata level (32.9 – 23.9% compared with 21.8–18.7%), but overall forest cover remained similar (104.8% in Plot 1, 103.2% in Plot 2), and the % cover at the 100–600 cm strata level (32.9% – 23.9% in Plot 2, 21.8% – 18.7% in Plot 1) and >600 cm strata levels were somewhat higher in Plot 2. (Several of the new ants are normally found under twig and leaf litter, so these components of the forest floor must have been sufficiently plentiful in 2009 to encourage additional colony founding by queens of litter-loving species.) The recovery of two normally arboreal species (both in genus *Anonychomyrma*) in 2009 could best be interpreted as their colonies being more abundant and ground foraging strays being more likely to be taken in pitfall traps in 2009. What also should be taken into account is the recovery of the ant fauna an extra two years after recent burning. In all, Plot 2 reveals a healthy spread of ants from different behavioural niches despite the modest richness numbers.

Plot 3 reveals a significant dip in ant species richness in 2008 (from 33 to 24 species), and a return and reinstatement of large numbers of species in 2009 (35 species). No less than seventeen ant species were collected in 2006 and 2009 but not in 2008. A perturbation in the normal equilibrium of ant biodynamics is evident in the latter year, but this appears to have disappeared by 2009. Ants strongly connected with disturbed conditions include *Rhytidoponera metallica* (F. Smith), whose numbers rose from 48 individuals (2006) to 84 individuals (2008) and had declined to 38 workers by 2009. Heat loving *Melophorus insularis* Wheeler also recorded a much smaller peak (21→46→36) in 2008. The ants unique to 2008 (eight species from six ant functional groups) are too diverse for any clear-cut explanations to be offered, but some of these could have been responding to short-term effects from the opening up of the canopy and increase in leaf litter (38.7% in 2006 to 64.9% in 2009).

Plot 4 provides evidence of fairly stable ant richness but increased numbers of ants in 2009 (no sampling done in 2008). The greatest rises were seen in small

myrmicines such as *Monomorium sydneyense* complex sp. JDM 101 (completely absent in 2006 but represented by 161 workers in 2009) and *Monomorium sordidum* (16 workers in 2006 and 183 workers in 2009), and in heat loving species (three species of *Melophorus*), among which *M. ludius* Forel (41→192 workers) exhibited the greatest increase. The massive decrease in SPH (820 to 154), increase in litter cover (47.5% – 64.3%) and general decline in forest cover would suggest that these small generalist scavengers and larger thermophiles found ready pickings of prey and edible vegetative matter as a result of the thinning.

The loss of four Cold climate specialists (three *Stigmacros* and a *Dolichoderus*) in Plot 5 over 2008/2009 is significant, since this plot, alone among the control and treatment plots, was burned during the survey period. A fifth Cold climate specialist returned in 2009 but was apparently not present in 2008. *Stigmacros* sp. JDM 443 was collected only in 2008, but no Cold climate specialists appeared among the novelties in 2009, although *Stigmacros glauerti* McAreavey was recovered again, after being absent from the catch in 2008. Three dolichoderines were added to the species totals in 2008/2009 and a fourth one (*Anonychomyrma nitidiceps* André) was collected in 2008 only. Opportunists not previously captured (four species) were also well represented among the novelties in 2008/2009, as were small generalised myrmicines (three *Monomorium* from the *M. Monomorium* group) and a *Pheidole* (*Pheidole* sp. JDM 164). The thermophile *Melophorus insularis* was likewise collected only in 2008 and 2009. The loss of the Cold climate specialists and the advent of those groups of taxa normally associated with a bare, insulated ground surface is consistent with the very marked reduction in canopy cover (from 46.5% in 2006 to 20% in 2009) and overall forest cover (75.2% to 52.9%) experienced by this plot during the sampling period. Most vegetative strata values also declined during this period. From a monitoring perspective, this plot shows perhaps the clearest and most easily interpreted signs of an alteration in the composition of its ant fauna that has been driven by deliberative acts.

Plot 6 is rather aberrant in that only one species present in 2006, the apparently very localised *Meranoplus* sp. JDM 491 (known from a small area in the Darling Range southeast of Perth), was not collected subsequently. No fewer than 12 ant species were collected only in 2008, including the solitary predator *Leptogenys neutralis*, three small generalist *Monomorium*, two *Meranoplus*, two *Stigmacros* species and two cryptic species (both *Plagiolepis*). Considering that commercial thinning, at least, took place in late December 2007, this surprising efflorescence of ants that utilise multiple environmental niches is counter intuitive, and not easy to explain. Since the collection took place not long after thinning operations, ant activity may have been correlated with scavenging of insects displaced or killed as a direct or indirect result of thinning and the deposition of branches and other tree waste on the ground surface. By 2009, the ant richness had declined to a roughly similar level to that of 2006 (i.e., 21 species, compared with 18 species in 2006 and 27 species in 2008). The abundance was much higher however, mainly due to very high numbers of the small myrmicine *Monomorium sordidum* (which is presumably treated as 'Generalised Myrmicinae' in

Andersen's papers but should probably be regarded as an Opportunist). Ants present in both 2006 and 2009 included no less than three *Stigmacros* species and the cryptic *Brachyponera lutea* (Mayr), which suggests some restoration of ground cover by the end of the survey period. Six ants were collected only in 2009, of which only two (*Anisopheidole antipodum* (F. Smith) and *Meranoplus ferrugineus* Crawley, both of which are Hot climate specialists) occupy the same ant functional group. The increase in functional groups that prefer to nest under cover and cool conditions may be partially explained by the increase in log density in the plot. *Stigmacros* species and *Brachyponera lutea* are commonly found in litter or under rocks or logs.

CONCLUSIONS

In summary, the various shifts in ant dynamics do not present a clear pattern over all the plots. Probably the change in equilibrium between ant ecological groupings noticed in plot 3 in 2008 and the loss of cover-loving species in Plot 5 and their replacement by those that favour open ground are most easy of interpretation. Plot 5, where burning was conducted, could presage the replacement of an ant fauna that was adapted to cool, moist conditions by one that favours warmer conditions and of under-cover nesters with open soil nesters. Potentially, permanent loss of some species and a depauperisation of the ant fauna generally could eventuate. By the same token, biological signals of serious degradation of the environment (such as drastic rises in abundance of 'weedy' dominants such as *Iridomyrmex chasei* Forel and meat ants, or the advent of exotics like *Pheidole megacephala* (Fabricius) or the Argentine ant *Linepithema humile* (Mayr)) appear to be completely absent from the survey sites.

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Appendix 1. Vegetation characteristics of plots (Adapted from Kabay, 2006, section 4.3).**VEGETATION CHARACTERISTICS OF PLOTS****NON-COMMERCIAL THINNING**

Treatment Area 1, Coupe 3

SITE 1, off Chandler road

418107E, 427780N thence NNW, line 2 417957E, 427957N

Basal areas- 34, 38, 44, 20 Average= 34m²/ha

Jarrah dominant regrowth, cut-over forest with few Marri, to 35m, with very few commercial saw log trees. Allocasuarina, Banksia, Persoonia, Balga present. Burnt a long time ago, continuous ground fuel with deep skirts on Balgas. Last burn 1987–88, fuel age 19yo. Dwellingup surface, with some lateritic outcrops, possibly Havel P/S or P type. Access is by good gravel road. Clean-down using brush only. Near bauxite rehabilitation.

SITE 2, off Chandler road

418778E, 428389N thence W, line 2 418772E, 428337N

Basal areas- 36, 28, 22, 22 Average=27m²/ha

Jarrah dominant regrowth, cut-over forest with few Marri, to 35m, with very few commercial saw logs and a few large habitat trees. There are few Allocasuarina, and Banksia, but more Balga present. Burnt a long time ago, continuous ground fuel with deep skirts on Balgas. Last burnt 1988–89, fuel age 18 years old. Dwellingup surface, gravelly soils, possibly Havel S type. Access is by good gravel road. Clean-down using brush only. Near bauxite rehabilitation

COMMERCIAL LOGGING FOLLOWED BY NON-COMMERCIAL THINNING

Treatment area 2, coupe 1.

SITE 3, off Chinaman road

424352E, 424410N thence NNW, line 2 424212E, 424615N

Basal areas= 34, 46, 36, 36, Average= 38m²/ha

Jarrah dominant cut-over forest with some Marri, to 35m, with some large commercial saw log trees. Allocasuarina, Banksia, Persoonia, Balga present. Burnt some time ago, continuous ground fuel with deep skirts on Balgas. Burnt 2002–03, fuel age 4.5 years old. Dwellingup surface, with lateritic outcrops, possibly P or P/S type. Access is by reasonable gravel track, mostly in dieback and Jarrahdale road. Clean-down using brush only.

SITE 4, off Chinaman road

424621E, 423902N thence N, line 2 424580N, 424127N

Basal areas- 32, 46, 44, 42. Average= 41m²/ha

Jarrah dominant cut-over forest with some Marri, to 35m, with some large commercial saw log trees. Allocasuarina, Banksia, Persoonia, Balga present. Burnt some time ago, continuous ground fuel with deep skirts on Balgas. Burnt 2002–03, fuel age 4.5 years old. Dwellingup surface, with lateritic outcrops, possibly P or P/S type. Access is by reasonable gravel track, mostly in dieback and Jarrahdale road. Clean-down using brush only.

Appendix 2. Wungong Catchment Environmental data for ant monitoring periods in 2006 and 2009 (adapted from Kabay, 2006, 2009).

Date	Maximum air temperature (°C)	Minimum air temperature (°C)	Precipitation to 9 am (mm)
7-Mar-06	34.5	17	0
8-Mar-06	33.7	17.5	0
9-Mar-06	31	17.2	0
10-Mar-06	27.2	15	0
11-Mar-06	23	13	0
12-Mar-06	26.5	11.5	0
13-Mar-06	29.6	12.2	0
14-Mar-06	29.6	13	0
15-Mar-06	29.5	14	0
7-Mar-09	28	14.5	0
8-Mar-09	31.5	15.8	0
9-Mar-09	32.5	15.6	0
10-Mar-09	35.1	15.2	0
11-Mar-09	28.8	12.5	0
12-Mar-09	no data	15.5	0
13-Mar-09	no data	no data	0
14-Mar-09	no data	no data	0
15-Mar-09	no data	no data	0

BHL



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Vegetation cover and not size of remnants determines composition and diversity of ground-dwelling arthropods in native vegetation remnants

JOSLYN BERKELAAR^{1*}, PETER SPELDWINDE¹ & CLAIRE FARELL²

¹Centre of Excellence in Natural Resource Management, University of Western Australia, Albany, WA 6330 Australia

²University of Melbourne, Parkville, VIC 3010 Australia

*corresponding author: ✉ joslyn.berkelaar@uwa.edu.au

Human urban populations continually grow and expand around the globe, and the urban footprint can directly and indirectly have deleterious effects on biodiversity of native flora and fauna through fragmentation. This study examined whether remnant area and habitat type between urban remnants affected arthropod biodiversity. Eighteen remnants within urban areas of a growing city in the South-western Australian Global Biodiversity Hotspot were surveyed using pitfall traps for ground-dwelling arthropods. Contrary to our hypothesis that arthropod diversity would increase in larger remnants, we found that size of remnant habitats had no effect on arthropod diversity; rather habitat composition had a much greater influence on arthropod diversity. Although remnant size had no significant effect on arthropod diversity, larger remnants supported a greater diversity of species that utilise the same type of resources, known as functional guilds. In our study we found that phytophagous (herbivores) and parasitoid functional guilds were more abundant in larger fragments, while the habitat structure and cover in each remnant affected scavengers, detritivores and pollinators. The abundance of angiosperms in remnants increased arthropod pollinator diversity, while increased sedge (Cyperaceae) cover decreased pollinator diversity. Interestingly, an increase in tree and leaf-litter cover decreased the number of detritivores collected. As all sites were identified as “ecologically functional” with maintenance of biogeochemical cycling, this is likely to closely reflect the arthropod diversity in Albany’s remnants and would have outweighed the effects of remnant size on diversity. This concludes that healthy habitat patches of all sizes are useful to maintain arthropod populations.

KEYWORDS: Vegetation Composition; Arthropods; Functional Guilds; Urban Remnants; Island Biogeography; Habitat Matrix.

INTRODUCTION

Human populations are increasingly being concentrated in urban areas around the globe, with 70% of the world’s population projected to live in urban areas by 2050 (United Nations 2008). For example, in Australia, 83% of the population currently live in cities and towns (ABS 2012). As human populations are increasingly concentrated into urban areas, native vegetation is fragmented by development and infrastructure and remnant habitats in cities can be some of the last remaining examples of ecosystems that once covered vast areas of the landscape (e.g. Sydney’s Eastern Suburbs Banksia Shrub and Melbourne’s Western Basalt Plains Grasslands (McDonnell 2007)). Consequently, urban areas can become a mix of infrastructure, humanity and remnant ecosystems.

Urbanisation fragments native vegetation resulting in vegetation patches that range in size, shape and connectivity. Native vegetation is considered vital for maintaining ecosystem processes within urbanised environments as vegetation controls temperatures, erosion, water runoff, nutrient cycling, air quality and provides habitat for wildlife (Dodds, Wilson *et al.* 2008). Fragmentation of native vegetation and the introduction of urban infrastructure, such as roads and

drains modifies these processes (Alberti, Marzluff *et al.* 2003) and reduces the area available for remnant flora and fauna communities (Oliver, Hong-Wa *et al.* 2011). Urbanisation is considered a major anthropogenic risk to biodiversity (FitzGibbon, Putland, & Goldizen 2007) and the preservation of remaining habitat remnants has been identified as being critical for maintaining urban biodiversity (Bennett & Gratton 2012). To this end, ecological studies, worldwide, are establishing relationships between landscape structure, urban development and the persistence of native species (Alberti 2005).

Fragmentation and the introduction of barriers confines biota to isolated pockets or patches within an urban landscape, increasing the likelihood of species confined to only a few small remnants (Parker & Mac Nally 2002). For these species, the ability for species to move between disconnected habitats becomes vital for maintaining demographic and genetic stability of populations (Magle, Theobald & Crooks 2009). Additional influences on species survival in fragmented urban patches include their isolation, dispersal ability, ability to survive stochastic events and the habitat health of the remnant (Drinnan 2005; Niemela 1999). Species movement between patches is a function of their dispersal ability and patch isolation (Braaker *et al.* 2014; Magle *et al.* 2009). Urban development generally increases the unusable habitat for native species around habitat remnants (Bennett & Gratton 2012) resulting in the

isolation of remnant vegetation patches. The survival of a species, during stochastic events in small remnant habitat patches, may depend on its mobility and its ability to colonise other fragments and establish and maintain breeding population (Abensperg-Traun & Smith 1999).

Vegetation structure and the health of a patch will influence its biological value. The greater diversity of habitats within an area increases the number of species (Hortal, Triantis *et al.* 2009). Community composition of a patch tends to be altered by urbanisation through the introduction of 'edge effects' (Marcantonio, Rocchini, Geri, Bacaro & Amici 2013; Porensky & Young 2013). Edge effects change the natural patterns of wind, light, temperature across the landscape (Porensky & Young 2013) and can also facilitate the introduction of exotic weed and pest species (Bolger, Suarez *et al.* 2000); the effect of which is expected to be greater in smaller and more irregularly shaped remnants (Porensky & Young 2013). Consequently, urbanisation often results in simplification of vegetation structural diversity within patches (Bryne 2007). Therefore, maintaining vegetation structure and health will provide a diversity of habitat types and have a positive influence on species survival (Cook 2002) and contribute to maintaining key biophysical processes such as: preventing soil erosion, reducing flooding and protecting water quality (Naiman & Decamps 1990).

To understand the effects of urbanisation and fragmentation on ecosystem functioning, ground-dwelling arthropod diversity and composition have often been studied (Bennett & Gratton 2013; Kowarik 2011; McIntyre, Rango, Fagan & Faeth 2001; Philpott *et al.* 2014). Arthropods provide a useful model for investigating the effects of urbanisation as they represent the most diverse taxon in most ecosystems and are vitally important to the health of the natural environment (Bolger *et al.* 2000; McIntyre 2000). In ecosystems, ground-dwelling arthropods help break down and redistribute nutrients into the soil (Didham, Ghazoul, Stork, & Davis 1996; Bolger, Suarez *et al.* 2000; McIntyre, Rango *et al.* 2001), pollinate flora (Didham, Ghazoul *et al.* 1996), biologically control the rate at which plants and pest species grow and multiply (Bolger *et al.* 2000; Bennett & Gratton 2012) and are the basis for many food webs (Bolger *et al.* 2000). In urban landscapes the relationship between arthropod diversity and fragmentation characteristics such as fragment size and connectivity is not a linear relationship but instead dependent on thresholds or minimum remnant size (Drinnan 2005). Urbanisation alters arthropod abundance and diversity through changes in land use, habitat structure and climate (Bennett & Gratton 2013; McIntyre *et al.* 2001; Philpott *et al.* 2013). For example, Gibbs and Hochuli (2002) found that anthropogenic disturbances such as habitat fragmentation alters arthropod assemblages, with opportunistic species, particularly spiders and wasps from higher trophic levels, becoming more common in smaller habitats. In terms of land use change, Bennett and Gratton (2012) found that changing from rural to urban land use negatively affected parasitic Hymenoptera abundance and diversity. In this same study, parasitoid abundance increased at a local scale as the floral diversity increased within urban sites (Bennett & Gratton 2012).

Conserving biodiversity within our cities is a global issue that commonly focuses on establishing protected natural areas and linking corridors (Hostetler *et al.* 2011). To assess whether urban remnant planning needs to be based on size of remnants or on the vegetation composition of each remnant, the effects of both patch size and habitat composition and health on arthropod diversity and abundance were examined in remnant native vegetation patches. This study focused on habitat patches in a growing regional city situated in south-western Australia, within a global biodiversity hotspot. We hypothesized that: 1) the size of vegetation remnants would have a significant positive influence on the ground dwelling arthropod composition and abundance; and 2) more diverse vegetation structure and composition would increase the arthropod diversity in remnants.

METHODS

Site description and selection

Vegetation structure and arthropod biodiversity of individual remnant patches within the City of Albany, Western Australia (population approximately 35,500 [City of Albany 2007]) was investigated. The City of Albany is a growing regional centre situated within the South-western Australian Biodiversity Hotspot. By the beginning of the 21st century, 63% of the vegetation within the City of Albany local government area had been cleared for agriculture and urban growth (City of Albany 2007). Within the Albany region over 800 flora species have been recorded in a variety of vegetation types (Heath, Low Woodland, Scrub, Reed swamps, Woodland/Forest, Scrub-Heath) (Sandiford & Barrett 2010). A total of 38 reserves within the city boundaries had been established for the preservation of vegetation, historical values and significant wetland habitats. From the 38 reserves in the City of Albany, we excluded wetland reserves from our study and those reserves sharing common boundaries were amalgamated; restricting the total number of reserves in the study to 18 (Figure 1). The majority of the landscape between reserves was urban residential, commercial and light industrial.

Vegetation structure, composition and Landscape measures

The 18 remnant reserves ranged in size from 0.26 – 219.8 ha, and were then classified into three arbitrary size categories 'large' (> 75 ha), 'medium' (6–74 ha), and 'small' (< 5.5 ha). The area (ha) of each reserve was calculated using ArcGIS (ESRI, version 8). Remnants were also classified initially into four distinct habitat types based on vegetation structural composition following Specht's (1970) classification; granite outcrop, shrubland, heathland, forest. In order to classify the structure and composition of vegetation within each of the 18 vegetation remnants, Landscape Function Analysis (Tongway & Hindley 2005) and the Landscape Organisation Index (Tongway & Hindley 2005) were used. Landscape Function Analysis (LFA) indirectly determines ecosystem function and resource capture by calculating the proportion of patch (indicating potential resource capture) to inter-patch (indication

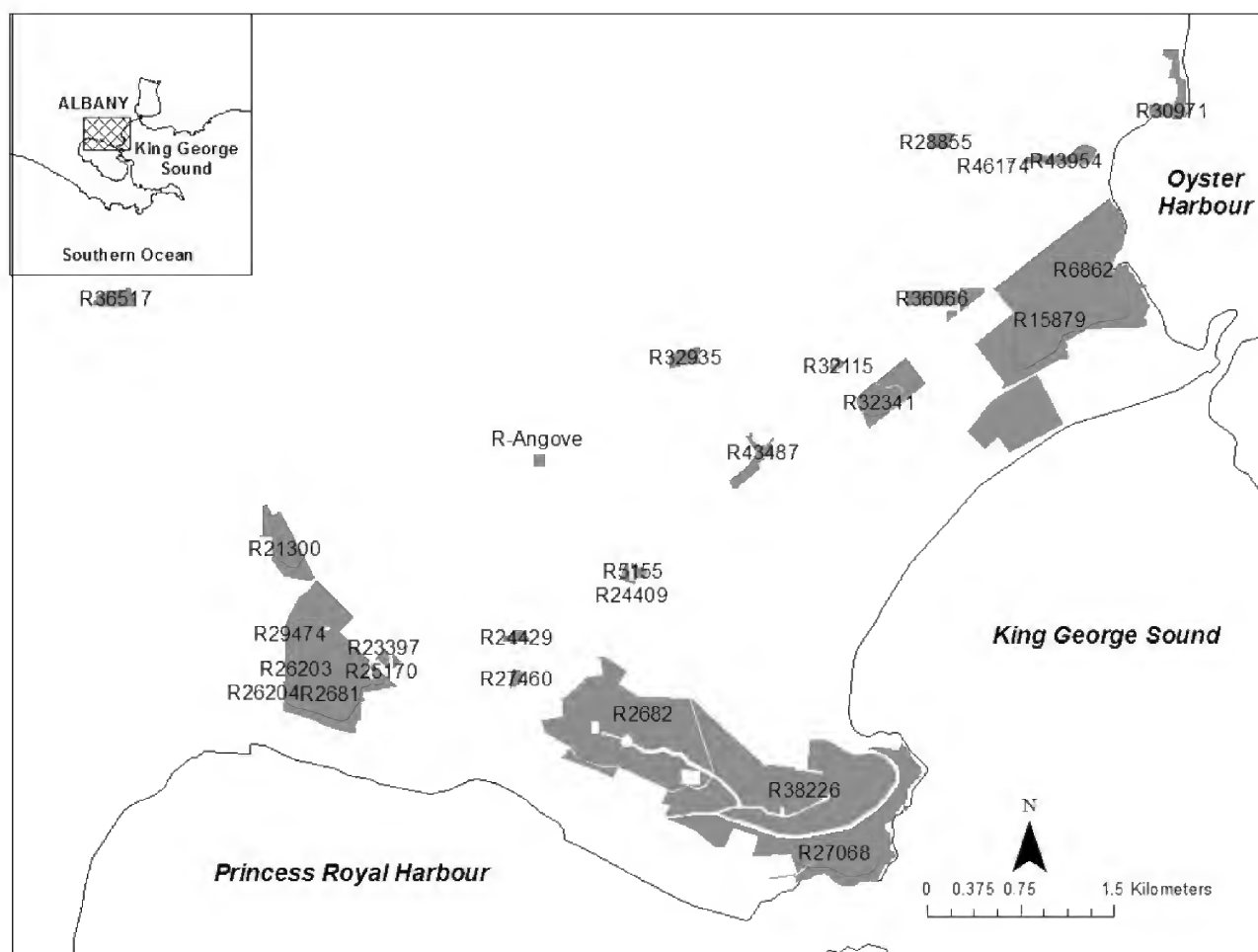


Figure 1. Remnant Bushland reserves in the city of Albany, WA.

potential resource loss) areas along transects. These simple indicators (patch and inter-patch zones) have been used to assess how well an ecosystem works as a biogeochemical system, i.e. whether resources are lost or retained by the system (Ampt 2007). It has been used around Australia on agriculture, orchards, and rangelands and provides useful comparison between different land-uses, on similar land types (Ampt 2007). In this analysis, patches are landscape features in contact with the ground (e.g. rocks, leaf-litter, low shrubs, stems, trunks and tussocky grasses), which prevent loss of water and nutrient resources from the system (Ludwig & Tongway 1997). Landscape Function Analysis determines the percentage cover of all vegetative life-forms as well as the entire patch and inter-patch zones along each transect. The patch and inter-patch data is then used to calculate a Landscape Organisation Index (Tongway & Hindley 2005).

The Landscape Organisation Index was calculated as the sum of all patch lengths along the transect divided by the total length of transect. The index places a landscape site on a continuum between highly functional where all resources are likely to be retained ($LO = 1$) and dysfunctional where resources are lost ($LO = 0$) (Tongway & Ludwig 1997). For example, a totally bare transect would have an index of 0 (zero) or if the transect was

entirely covered by resource trapping patches (e.g., forest with closed understorey or continuous leaf-litter cover) the index would be 1.

For determination of the Landscape Organisation Index data was collected along two 20 metre long transects established in the centre of each fragment (Tongway & Hindley 2005). Location in the centre of remnants minimised edge effects which may have confounded interpretation of the data (Bolger, Suarez *et al.* 2000). Each transect was positioned parallel to the slope, in the observed direction of surface water flow and the location and orientation of each transect was recorded. All vegetative life-forms that intersected transects on a vertical plane were recorded. The length (extent of vegetation along the transect tape measured in metres), height (m) and percentage cover (1 m² quadrats) of vegetation patches (tree, shrub, herb, weed, grass and sedge), and the length (m) and percentage cover (1 m² quadrats) of non-vegetation groundcover patches (cryptogam, woody debris and leaf-litter) were measured. The length (extent along the transect tape, metres) of bare ground and rocks defined as inter-patches was also recorded. One person collected all estimated percentage cover (canopy and non-vegetation components) visually to avoid multi-observer bias in the measurements.

Arthropod Collection

Three pitfall traps (500 ml plastic specimen jars filled with 2.5 cm of ethylene glycol) were established along each transect (i.e. 6 per site) Gibb & Hochuli (2002). Traps were positioned at 0, 10 and 20 m from the origin of each transect (McIntyre *et al.* 2001). All traps remained closed for at least a week after deployment of trap lines to ensure arthropods were trapped under the same conditions for all transects and to avoid digging-in effects (Gibbs and Hochuli 2002). Traps were open for seven days during March, 2011. At the end of this period, samples were preserved in ethyl alcohol (70% ethanol) for later identification. Samples were sorted with the aid of a stereo dissecting light microscope and all arthropods and accidental captures were counted and identified to Family or Order level (CSIRO 1991), then morphospecies; differentiating arthropods within orders due to their external appearance, and functional guild. Spiders were identified to family using Schimming (2010); and ants were identified to subfamily using Shattuck (1999) and functional guild using Andersen and Majer (2004).

Statistical Analysis

Univariate data analysis was conducted using Genstat 12.1.0 (2009, VSN International Ltd.) and multivariate data analysis using Primer 6 (2001, Primer-E Ltd). Initially, remnants were characterised by area, habitat classification (Specht 1970) and Landscape Organisation Indices. A univariate approach was used to compare all vegetation characteristics (cover and vegetation life-form) observed among remnant area and habitat type. Differences in Landscape Organisation indices among remnants were determined using Analysis of Variance (ANOVA) and Tukey's studentised test (significance level set to $\alpha = 0.05$). Using the Bray-Curtis Distance Matrix; Principle Component Analysis (PCA) was used to examine patterns in vegetation composition and canopy

cover among sites. Arthropod data was pooled from all traps along both transects in each site. The richness and diversity (Shannon-Weiner Index) of both species and functional groups of arthropods among remnants was calculated. The Shannon-Weiner Index was calculated to take into account the species richness and the proportion of each species within a study site, as both of these factors influence diversity. All arthropod data were Log ($x+1$) transformed to satisfy assumptions of normality and heteroscedasticity. A similarity matrix between samples was constructed using the Bray-Curtis Similarity Index for community structure based on arthropod diversity.

The algorithm SIMPER (Primer) was used to identify the species and vegetation variables that contributed most to the similarities and difference between remnants. These variables were then analysed univariately to determine whether they had any significant influence on the functional guilds found in each of the remnants (regression analysis and ANOVA). The relationship between vegetation and biodiversity composition was examined using linear regression and ANOSIM to calculate whether there were any significant differences between patch vegetation structure or quality (habitat classification, vegetation characteristics and Landscape Organisation Index) and arthropod richness and diversity.

RESULTS

Habitat Data

The 18 remnants varied in size, with three 'large' (> 75 ha), 3 'medium' (> 6 ha), and 12 'small' (< 5.5 ha) patches. Seven habitat classes were identified; Closed Forest, Low Forest, Low Open Forest, Woodland, Shrubland, Granite Outcrop, and Heathland.

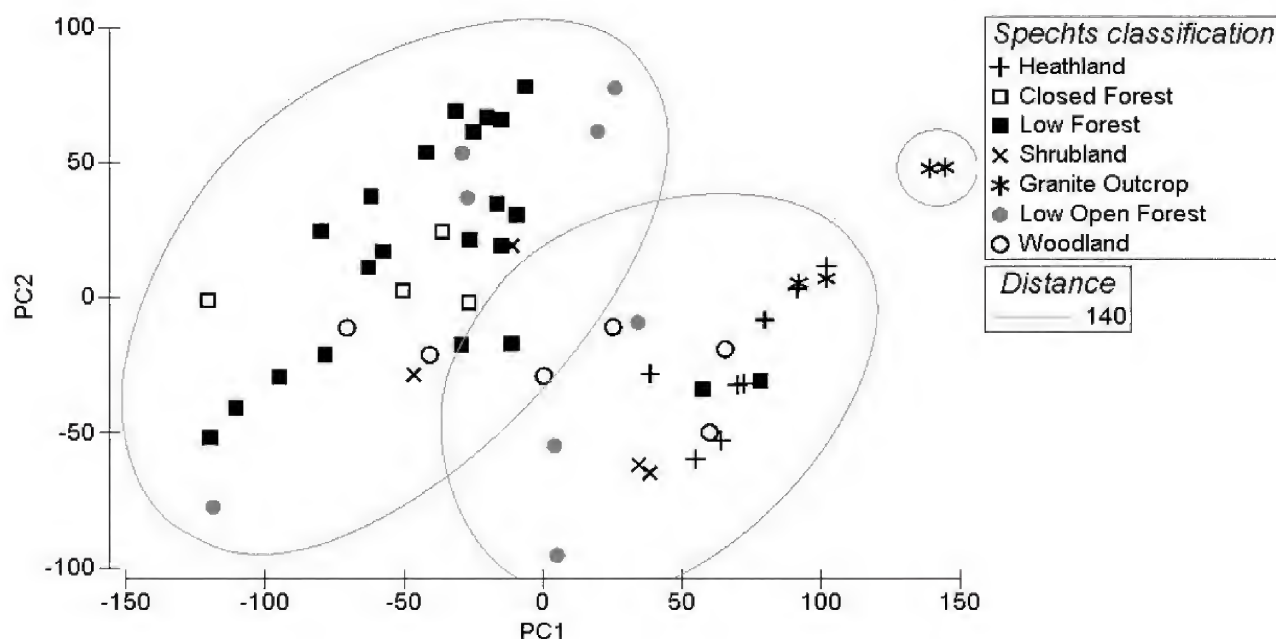


Figure 2. Principle Component Analysis (PCA) of remnants based on vegetative classification (Specht 1970) and objective clusters (circles) using dissimilarity matrix at Level 140.

Heathland and Granite Outcrop. Landscape Organisation indices for these habitat classes ranged from 0.48 (Granite outcrop) to 1.0 (Closed Forest). For the remaining habitat classes Landscape Organisation Indices were all greater than 0.78: Heathland (0.78), Low Forest (0.97), Low Open Forest (0.92), Woodland (0.92) and Shrubland (0.95). Remnant area had no significant effect on the Landscape Organisation Index of patches (ANOVA; $P > 0.05$)

Principle Component Analysis (PCA) identified substantial overlap in vegetation cover and life-form among the habitat classes identified using Specht's (1970) classification (Figure 2). The first two principle components together explained 72.7% of the variation among sites. Objective classification using dissimilarity matrix at level 140 combined habitat classifications into three broad vegetation structural types. Group 1: High Vegetative Structure comprised Closed Forest, Low Forest, and most Low Open Forest; Group 2: Low Vegetative Structure comprised predominately heathland, shrubland, and most woodland; and Group 3: Granite Vegetative Structure comprised solely of granite outcrop habitat (Figure 2). Of the 11 vegetation life-form or cover variables measured, PCA showed that five variables (percentage cover of canopy, trees, shrubs, leaf-litter and sedges along each transect) had the greatest influence on differences between sites (Table 1).

Arthropod Data

In this study a total of 10 352 arthropod individuals were collected, representing 163 morphospecies, 29

Table 1. Eigenvalues for Component 1(PC1) and Component 2 (PC2). The highlighted values are those that contributed most to the differences between remnants (Figure 2)

Variable	PC1	PC2
Canopy cover	-0.722	-0.436
Tree	-0.423	0.346
Shrub	-0.226	-0.308
Weeds	-0.002	-0.008
Herb	-0.042	0.029
Cryptogram	0.132	-0.150
Woody debris	-0.103	-0.007
Leaf-litter	-0.447	0.428
Inter patch	0.131	-0.015
Sedge	0.039	-0.623

orders and 9 functional groups (Table 2). The most diverse order was Hymenoptera with 37 different taxa, and the most abundant species was *Collembola Entomobryidae* sp.. Species diversity did not vary between remnants of different size ($P = 0.7$) (Figure 3), nor was there a significant difference due to ecological function (Landscape Organisation Index) of remnant sites (ANOVA, $P = 0.98$) (Figure 4). Although total species diversity did not vary between remnants of different sizes, when considered separately, members

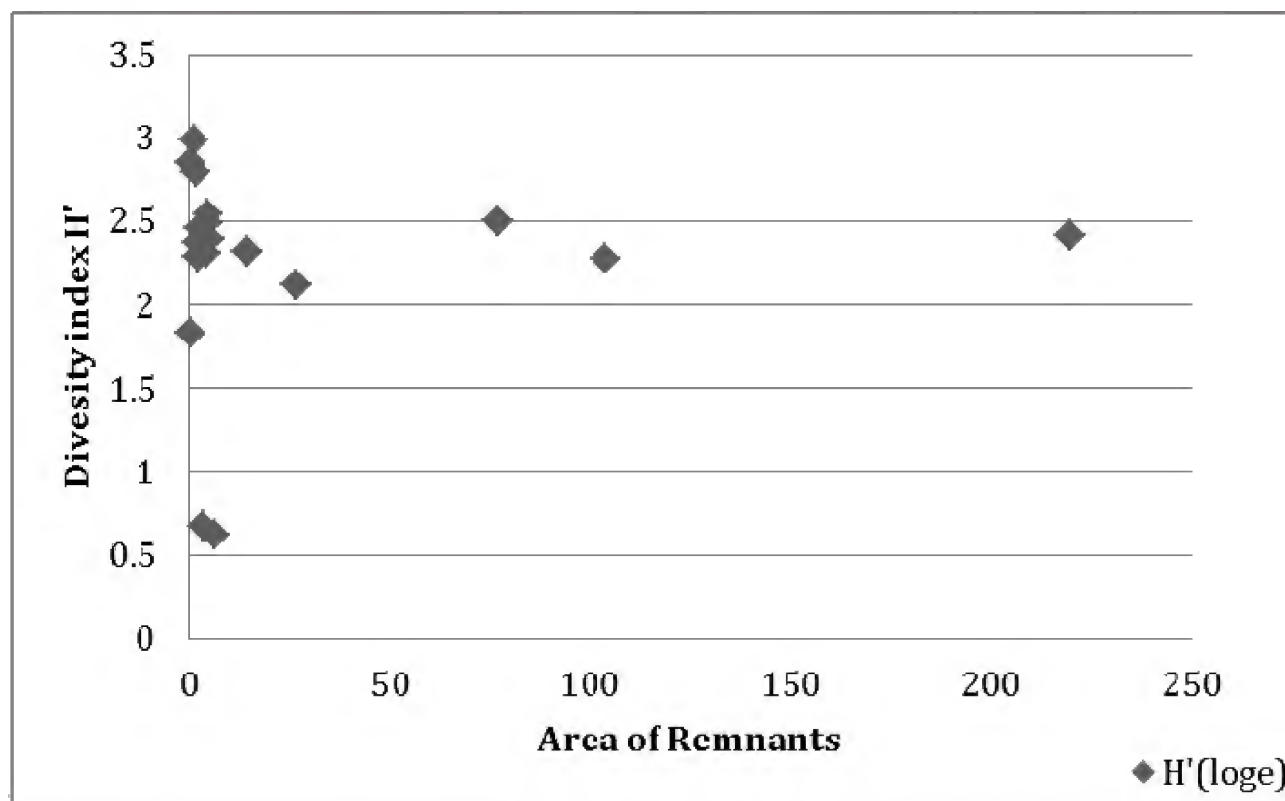


Figure 3. Effect of remnant size on diversity of arthropods in remnants surveyed within the City of Albany, Western Australia; $R^2 = 0.007$ showing that area has no effect on arthropod diversity.

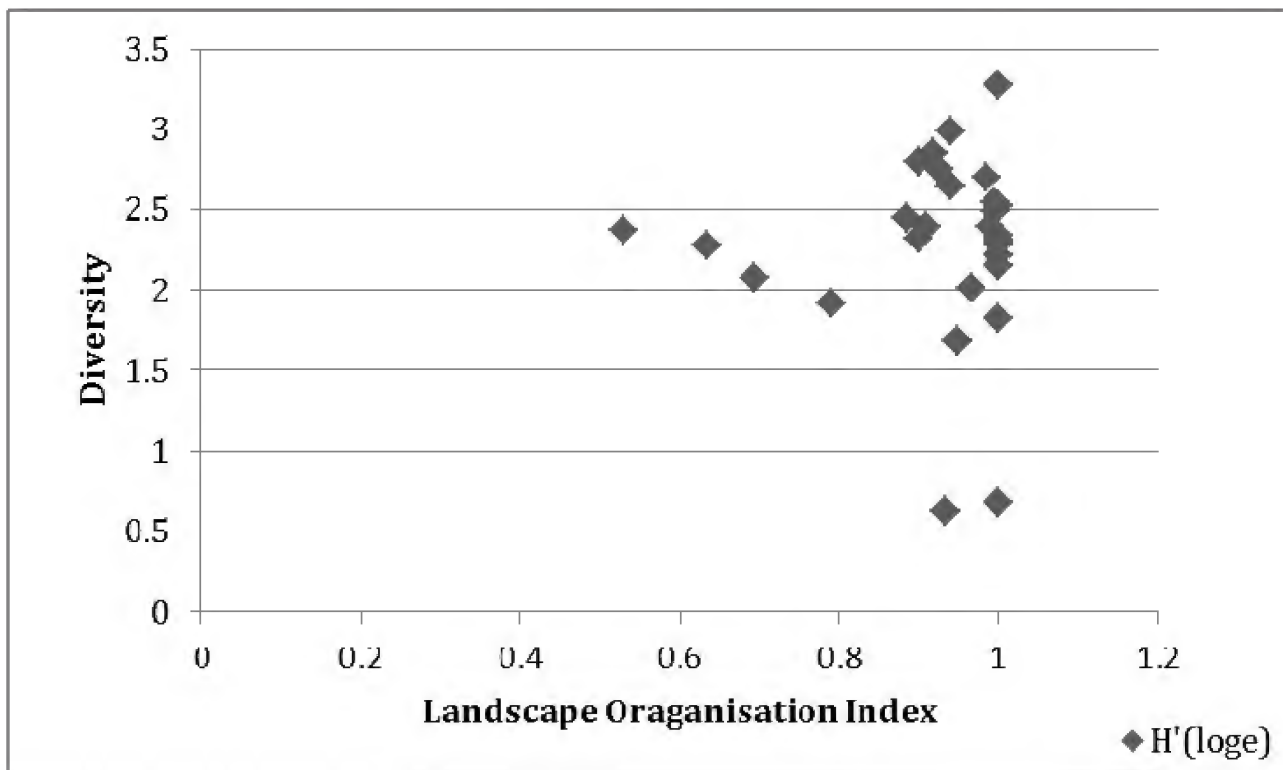


Figure 4. Relationship between Landscape Organisation Index (LOIndex) and diversity of Arthropods in remnants surveyed within the City of Albany, Western Australia ($R^2 < 0.01$).

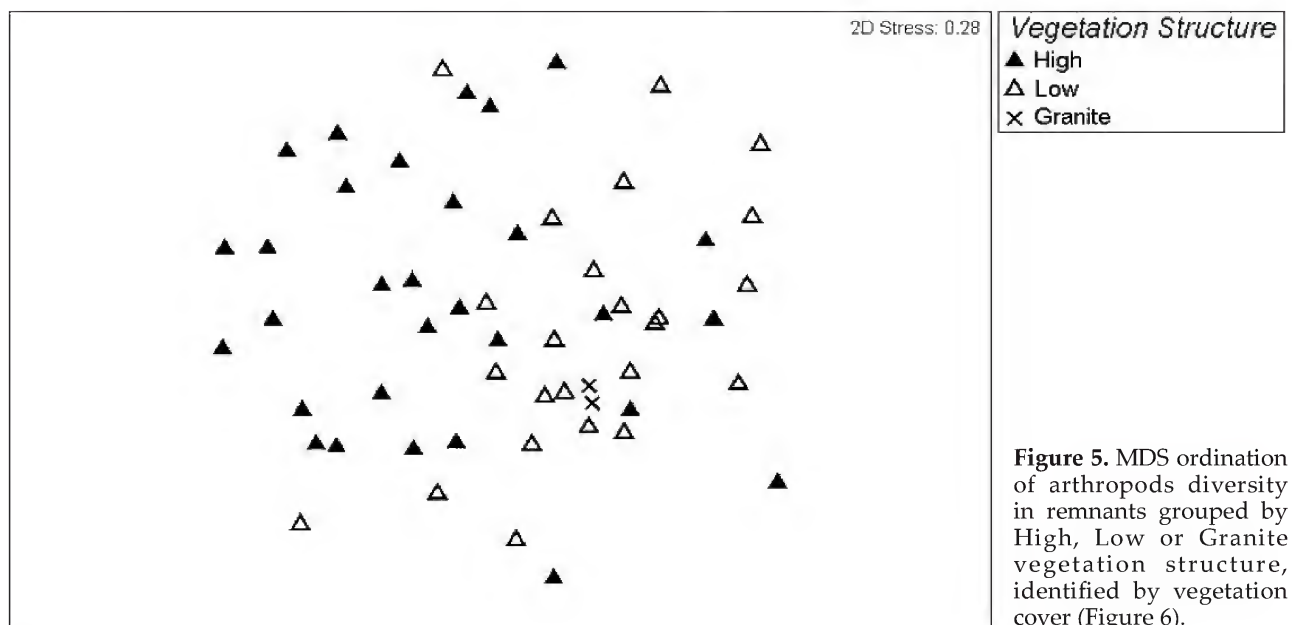


Figure 5. MDS ordination of arthropods diversity in remnants grouped by High, Low or Granite vegetation structure, identified by vegetation cover (Figure 6).

of phytophagous and parasite functional guilds were significantly more abundant in larger remnants (ANOVA; $P = 0.03$ and 0.02 , respectively).

There was no relationship found between species richness and Landscape Organisation Indices ($P > 0.05$) nor was there any difference in richness due to area of remnant ($P > 0.05$). There was no significant difference in arthropod diversity among habitat classifications (ANOVA; $P = 0.7$), and only weak differentiation among

vegetation structural groups (ANOSIM; Global $R = 0.086$, $P = 0.009$) (Figure 5). The species which contributed most to the similarities and differences within habitat groups were: Collembola, Formicidae and Amphipoda (SIMPER analysis).

The vegetation life-form and cover variables with the greatest influence on vegetation structural or habitat differences (Table 1) showed a non-significant influence on arthropod species diversity (Figure 5) but had a

Table 2. Arthropod Orders collected organized according to functional guilds.

Arthropod Orders	Predator	Phytophagous	Scavenger	Parasite	Detritivore	Coprophagous	Omnivore	Pollinator	Forager
Araneae	14	0	0	0	0	0	0	0	0
Acarina	22	0	0	0	0	0	0	0	0
Amphipoda	0	0	1	0	0	0	0	0	0
Blattodea	0	0	2	0	0	0	0	0	0
Chilopoda	1	0	0	0	0	0	0	0	0
Coleoptera	5	9	5	0	3	1	0	0	0
Collembola	0	0	0	0	3	0	0	0	0
Dermaptera	0	0	0	0	0	0	3	0	0
Diplopoda	0	0	1	0	0	0	0	0	0
Diptera	1	0	0	3	0	1	6	2	0
Formicidae	0	0	0	0	0	0	0	0	7
Hemiptera	0	7	0	1	0	0	0	0	0
Hymenoptera	2	0	0	3	0	0	0	25	0
Isopoda	0	0	0	0	1	0	0	0	0
Isoptera	0	0	0	0	1	0	0	0	0
Lepidoptera	0	4	0	0	0	0	0	2	0
Nematodiorpha	0	0	0	1	0	0	0	0	0
Neuroptera	1	0	0	0	0	0	0	0	0
Oligochaeta	0	0	0	0	1	0	0	0	0
Opiliones	0	0	0	0	0	0	3	0	0
Orthoptera	0	5	0	0	0	0	0	0	0
Phasmatoda	0	1	0	0	0	0	0	0	0
Phthiraptera	0	0	0	1	0	0	0	0	0
Pseudoscorpions	1	0	0	0	0	0	0	0	0
Psocoptera	0	0	1	0	0	0	0	0	0
Scorpion	1	0	0	0	0	0	0	0	0
Stephanocircidae	0	0	0	1	0	0	0	0	0
Thysanoptera	0	0	0	0	2	0	0	0	0
Thysanura	0	0	0	0	0	0	1	0	0

significant impact on the functional guilds found within the sites. While scavenger abundance was greater in remnant habitats with greater total canopy-cover, tree-cover and leaf-litter (ANOVA; $P = 0.005$, 0.0004 , and 0.007 respectively), these variables had a significantly negative impact on the abundance of detritivores collected (tree-cover; ANOVA; $P < 0.001$) and leaf-litter (ANOVA; $P = 0.009$). Pollinators were more abundant in remnants with greater tree-cover (ANOVA; $P = 0.05$) but decreased in those habitats with greater sedge-cover (ANOVA; $P = 0.02$). An increase in shrub-cover along transects had a positive significant influence on parasitic (ANOVA; $P = 0.008$) and coprophagous guilds (ANOVA; $P = 0.006$).

DISCUSSION

This study focused on the biodiversity of arthropods found in urban remnants within a growing urban centre. Contrary to our hypothesis that arthropod diversity would increase in larger remnants, we found that size of remnant habitats had no effect on arthropod diversity.

Although arthropod diversity has generally been found to increase in larger remnants (Bolger *et al.* 2000; Faeth & Kane 1978; Yamaura *et al.* 2008) other studies have also found no relationship between species richness and urban fragmentation (Gibb & Hochuli 2002; Oliver *et al.* 2011; Parker & MacNally 2002). For example, Parker and MacNally (2002) found that grassland invertebrates in south-eastern Australia did not respond to habitat loss or habitat fragmentation when mowing decreased available habitat by 60 and 90%. In our study, although we found that arthropod diversity did not increase, phytophagous (herbivores) and parasitoid functional guilds were more abundant in larger fragments, while the habitat structure and cover in each remnant affected scavengers, detritivores and pollinators. Gibb and Hochuli (2002) also found that species richness was not greater in large than small fragments, rather assemblage composition responded to fragmentation. They found that generalist species were more abundant in smaller remnants, while predators and parasitoids were negatively affected by fragmentation (Gibb & Hochuli 2002). On the other hand Christie, Cassis and Hochuli (2010) found that trees in larger habitat patches supported fewer arboreal

arthropods compared to tree in edges and small urban remnants. They also found a shift in functional grouping, with a greater number of herbivore invertebrates in small remnants and edge sites than interior sites (Christie *et al.* 2010).

There are a number of possible reasons for the absence of size effects on arthropod diversity in our study; firstly, the critical reserve size for arthropods could potentially be smaller than the remnant areas surveyed or that the number of remnants in each size class might not have been large enough to detect differences in arthropod diversity. Yamaura *et al.* (2008) surveyed 48 sites ranging in size from 2.4 to 296 ha and Bolger *et al.* (2000) studied 40 remnants ranging in size from 0.3 to 296 ha, while our study surveyed 18 remnants ranging in size from 0.25 to 219.8 ha. Another reason for the absence of any significant effect of remnant size on arthropod diversity in our study, is that, in the city of Albany, terrestrial remnant vegetation 'islands' are mostly surrounded by domestic gardens and green verges, which are landscapes that can be potentially utilised by arthropods as corridors (McIntyre 2000), essentially negating island and isolation effects as conceived in the Theory of Island Biogeography set out by MacArthur and Wilson (1963 and 1967). Key findings from a range of studies have found that the floral diversity and structural complexity of domestic gardens is an important predictor of arthropod abundance and diversity (Goddard *et al.* 2010).

Another more important factor that may have removed the effect of remnant size on arthropod diversity is ecosystem function of remnants. In our study, we used Landscape Function Analysis (LFA) to indirectly determine ecosystem function and resource capture by calculating the proportion of patch (resource capture) to inter-patch (resource loss) zones (Landscape Organisation Index). While Landscape Organisation Indices can range from 0.0 – 1.0 (Tongway & Hindley 2005), values in our study ranged from 0.48 – 1.0. This shows that for almost all the sites studied, that resource capture is greater than resource loss, implying that nutrients and water are conserved by the native vegetation rather than being lost from the system. As all sites were 'ecologically functional' with maintenance of biogeochemical cycling, this is likely to closely reflect the arthropod diversity in Albany's remnants and would have outweighed the effects of remnant size on diversity (McKinney 2008).

Although remnant size had no significant effect on arthropod diversity, larger remnants supported a greater diversity of species that utilise the same type of resources, known as functional guilds (Gardener, Cabido *et al.* 1995). In particular, larger fragments supported a greater abundance of phytophagous and parasitic arthropods. Larger fragments support greater vegetation, which increases the abundance of phytophagous arthropods (Bennett & Gratton 2013). As birds and mammals are more diverse in larger habitat fragments (Drinnan 2005; FitzGibbon, Putland *et al.* 2007) this would also increase the number of suitable hosts for arthropod parasites. Increased parasite abundance was also associated with increased shrub-cover along transects. Further, shrub-cover could also have increased the number of flowering species in these remnants and consequently the number of suitable host bird and mammal species for parasitoid arthropods. These

results were also found by Bennett and Gratton (2012) who found a positive relationship between parasitoid abundance and flower diversity.

Vegetation structural diversity is generally considered important for maintaining the health of native vegetation (Cook 2002; Brodie 2003). Structural diversity also increases the possibility of greater habitat diversity; likely yielding increased diversity and greater possibilities for species survival (Cook 2002). Studies have shown that ground-dwelling and soil-arthropods are strongly influenced by habitat structure (Bryne 2007), for example Loyola and Martins (2008) found a positive correlation between structural heterogeneity, tree abundance and shrub height and Hymenoptera richness and abundance. Floral diversity particularly increases resources for pollinator arthropods (Hodge, Marshall *et al.* 2010). Jaganmohan, Vailshery and Nagendra (2013) found that in Bangalore, India the number of insect orders increased as the number of tree, herb and shrub species increase. In our study, vegetation structure greatly affected pollinator abundance, with increased tree-, shrub-, and herb-cover increasing pollinator abundance, while increased sedge-cover decreased pollinator abundance. This is not surprising as most species of trees, shrubs and herbs within the studied remnants were flowering plants (angiosperms) and other studies have found arthropod pollinators are predominately found on angiosperms, which provide them with food resources (Herrera & Pellmyr 2002). Sedges on the other hand, are mostly wind pollinated (Herrera & Pellmyr 2002) and an increase in sedge cover relative to angiosperm cover could lead to a decrease in diversity of pollinating arthropods, due to decreased food supply. This is consistent with other studies, for example, Hennig and Ghazoul (2012) found that floral abundance has a positive effect on bee diversity and Potts, Vulliamy, Dafina, Ne'eman and Willmer (2003) found that floral richness is highly correlated with bee species richness.

Vegetation cover and the presence of leaf-litter also influenced functional guild abundance. While other studies have found that detritivores increase in forest habitats (Bryne 2007); increased tree- and leaf-litter cover along transects in our study significantly reduced the number of detritivores. An increase in total vegetation canopy-cover, tree-cover, and leaf-litter cover was also associated with an increase in the number of scavengers, which also live in the leaf-litter feeding on dead plant material and animals (CSIRO 1991). As detritivore abundance is related to leaf-litter and soil moisture content, a decrease in detritivores collected may also be due to the lower than average annual rainfall that Albany has received in the 3 years prior to our study (< 700 mm compared to the expected average of 930 mm) (Bureau of Meteorology 2011). Norton, Thomson, Williams and McDonnell (2014) also found that dry microclimate conditions had an effect on the arthropods collected, with more arthropods collected in grassland plots than bare patches. An increase in shrub-cover was accompanied by an increase in the numbers of coprophagous, or dung eating (Angel & Wicklow 1975), arthropods. As hypothesised for parasitoid numbers, increased shrub-cover may increase the number of vertebrates that live in or visit these remnants, increasing the food supply for coprophagous arthropods.

There are several conclusions that can be drawn from this study of remnants within a growing urban centre; firstly, the size of remnant had no significant effect on either ecological functioning or on arthropod or functional guild diversity. This lack of effect could be due to the fact that our remnant scale was either too big or too small to observe any significant differences or due to the high ecological functioning of these remnants; although additional studies should be carried out to test this further. Secondly, arthropod diversity and presence or absence of functional groups was highly influenced by vegetation structural composition within remnants, with percentage total canopy-, tree-, shrub-, sedge- and leaf-litter- cover being the most important habitat variables determining diversity.

Consequently, in urban areas where vegetation remnants are highly functional, a diversity of vegetation life-forms and habitat structure may be more important than remnant size for maintaining arthropod diversity. This may lead to planning in urban areas to focus on the maintenance of habitat health and structural diversity when planning their green areas and corridors for native biodiversity.

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Ecology and Distribution of the Slider Skink, *Lerista neviniae*

GLEN GAIKHORST

GHD Pty, Ltd. GHD House 239 Adelaide Terrace, Perth, WA, 6004

✉ Glen.Gaikhorst@ghd.com

Abstract

Many *Lerista* are described based on few individuals, often with little or limited descriptive information which consequently classifies them as data deficient. These species tend to be researched or observed by specialist herpetologists or as part of environmental approval processes for mining or infrastructure development. *Lerista neviniae* was described from three individuals by Smith and Adams (2007) and subsequently listed under the Western Australian *Wildlife Conservation Act 1950* as Vulnerable due to the very small distribution for the species and the increasing pressure on its habitat.

The species is only known to occur in the Cape Lambert area on the north-west coast of Western Australia. This paper presents additional recorded specimens, habitat usage, distribution and potential impacts for the species.

KEYWORDS: Pilbara, Cape Lambert, *Lerista neviniae*, vulnerable, habitat, distribution, impacts

INTRODUCTION

L. neviniae is a small fossorial skink that is very pale with prominent black paravertebral and upper lateral stripes. The skink has four limbs, each with three digits and 18 midbody scale rows (Smith and Adams, 2007; Wilson and Swan, 2013). Due to its pattern it is the most distinctive member of the recently revised *Lerista muelleri* species complex (Smith and Adams, 2007). *L. neviniae* is shown in Plate 1.

The species was described using three specimens. The holotype (Western Australian Museum 135295) was collected in 1997 east of Pope's Nose Creek (20°36'45.6"S, 117°10'39.1"E; the type locality). Two paratypes (Western Australian Museum 151303, 135306) were additionally collected in fore dunes just east of the type locality and approximately 9 km south-west of the type locality. All collection locations are in the general vicinity of Cape Lambert in the Pilbara region of Western Australia. At the time of collection the species had a known distribution limited to several low coastal dunes, with initial estimates of the species' range being approximately 330 ha (Biota, 2008a). The vegetation in the area comprised *Acacia* spp. and other shrubs over *Spinifex longifolius* (Biota, 2008a). Results from hand foraging searches indicated the species utilises small logs or woody debris on white primary dunes of coarse sands (Smith and Adams, 2007).

Cape Lambert is a major export port servicing the mining industry in the Pilbara. Mining has expanded in the Pilbara region significantly in the last 20 to 30 years which has included expansions to the infrastructure in the Cape Lambert area and within the habitat of *L. neviniae* Biota (2008a, b, c, 2009), GHD (2010a, b, c, 2011), Ninnox (2008), Phoenix (2010), SKM (2009), and API (2009, 2010, 2011). The aim of this paper is to present new data obtained from field studies in the Cape Lambert area by consultants and private individuals which extends the known range of the species in the region.

METHODOLOGY

As part of environmental approvals processes, a number of targeted surveys has been conducted in the region for *L. neviniae*. Due to this survey effort, a broader area of dune habitat suitable for *L. neviniae* has been surveyed between Karratha and Sherlock Bay. The search areas were disbursed along approximately 70 km of low coastal dunes and associated sandy areas. Additionally, islands in the region have been searched for this species including Dixon, Delambre, Jarman, Legendre, Bezout and Pricard Islands.

Survey techniques included both hand foraging (raking) and pit trapping in sandy areas of suspected habitat of *L. neviniae*. Actual methodology of trap design varied between projects.

A number of unpublished reports and publications was available for review including environmental approval documents (listed above), Smith and Adams (2007), Wilson and Swan (2013) and Richardson *et al.* (2007). Additionally the author has conducted several private field trips to the region undertaking hand foraging surveys (raking) between Karratha and Sherlock Bay in sandy and dune areas within 5 km of the coastal strip. Trips have occurred between 2009 and 2013 for a period of 4–14 days. All *Lerista* specimens were identified using Smith and Adams (2007) and Wilson and Swan (2010, 2013).

Eighteen specimens of *L. neviniae* currently reside in the Western Australian Museum collection. These consist of 16 from the Cape Lambert area and 2 from Dixon Island. These specimens were sexed (15 individuals) and measured for Snout Vent Length (SVL) and Tail Length (TL).

RESULTS

Distribution

The distribution of *L. neviniae* was found to extend beyond the Cape Lambert area and to include the eastern side of



Plate 1. *Lerista neviniae* from eastern Anketell Point. (Photo: Glen Gaikhorst).

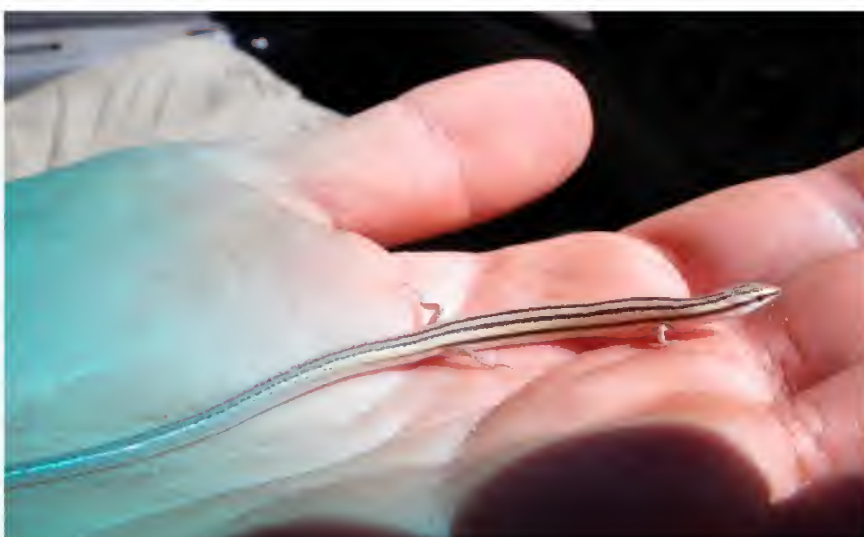


Plate 2. *Lerista neviniae* from Cleaverville Beach. (Photo by Bret Stewart).

Dixon Island. On the mainland *L. neviniae* has been found on the coastal fringe from Cleaverville Beach (furthest western point; see Plate 2) and the western side of Dixon Headland, almost to the end of Bouguer Passage around Anketell Point, Cape Lambert to near Point Samson. This includes patchily occupied areas of habitat along a 20 km stretch. At Point Samson, two individuals have been found approximately 2 km inland on isolated dunes. Currently, the known range of the species is approximately 608 hectares in size as shown in Figure 1.

Approximately 75 individuals have been captured. These are mapped below in Figure 2. (Note: multiple specimens have been captured at some sites and this is represented as a single point). Numerous records in the Department of Parks and Wildlife (DPaW) and Western Australian Museum *Naturemap* database have been excluded from this assessment due to inaccuracies.

Habitat

The type locality for this species was briefly described as coastal dunes vegetated by *Acacia coriacea* and low shrubs

(Smith and Adams, 2007). This description was later expanded to include pale coastal dunes with *Acacia* spp. and *Spinifex longifolius* (Biota 2008a). Further assessment includes dunes of suitable depth and excluding areas where rocks, gravel or shell fragments are present. One individual located on the eastern side of Anketell Point (See Plate 3) was found under man-made debris in primary white (pale) dunes with scattered *Acacia* spp. and small grasses. The animal was captured within 6 m of a mangrove inlet.

On the eastern side of Dixon Island the environment where several animals have been captured is slightly different. In this location the environment consists of fine white sands on low primary dunes with open *Acacia coriacea* Shrubland over *Cenchrus ciliaris* (Buffle grass) and *Aerva javanica* (See Plate 4). The site has very little surface debris and one specimen was captured under a dead clump of *Cenchrus ciliaris*. Specimens captured at Cleaverville Beach and near Point Samson were found in a very similar environment to that on Dixon Island, consisting of primary dunes with open *Acacia coriacea* Shrubland over *Cenchrus ciliaris* and *Aerva javanica*.



Figure 1. Habitat areas available to *L. neviniae* within known locations.



Figure 2. *L. neviniae* records. (Represented by triangles).



Plate 3. Habitat at Anketell Point. (Photo Glen Gaikhorst)



Plate 4. *Lerista neviniae* habitat Dixon Island. (Photo Glen Gaikhorst).

Two individuals were identified inland from Point Samson on what appears to be isolated dunes. These dunes have a darker sandy soil and lack the coastal vegetation previously identified. The dunes are dominated by *Cenchrus ciliaris* and this weed occurs in most of the sites visited in the Point Samson area.

Based on current knowledge of *L. neviniae*, the species' habitat consists of dunes within 2km of the coast of pale to dark, loose sand supporting scattered low shrubs of *Acacia coriacea* 10–30% and/or the introduced *Cenchrus ciliaris* and *Aerva javanica* over *Spinifex longifolius* 10–30%.

Museum Records

Eighteen individuals reside in the WAM of which 15 were measured and sexed. The remaining three were all measured but two remained too small to sex. Of the 15 samples ten were males and five were female. The SVL range for males is 34–40 mm and TL 20–55 mm. The average SVL is 37.8 mm and TL 40.3 mm. The SVL range

for females is 34–42 mm and TL 42–58 mm. The average SVL is 39.2 mm and TL 48.2 mm. These measurements show that female *L. neviniae* are slightly larger than males, however there is considerable overlap in the range between sexes. These measurements can be seen below in Table 1.

DISCUSSION

Based on the current knowledge of *L. neviniae*, it is likely to be restricted to coastal and near-coastal sand dunes from Cleaverville Beach to the northern side of Cape Lambert (near to Point Samson). It is possible that *L. neviniae* occurs on the remainder of Dixon Island (western side) in suitable habitat, however the species is likely to only occur in low numbers on the western side of Dixon Island given that targeted surveys have been undertaken in this area and no specimens were recorded. Between Karratha and Cleaverville Beach,

Table 1. *L. neviniae* retained at the WAM, sexed and measured.

WAM ID	Location collected	Sex	SVL	TL	Date collected
R135295 (Holotype)	Cape Lambert	Male	37	46	28/10/1998
R135306 (Paratype)	Cape Lambert	Female	40	42	27/10/1998
R150102	Cape Lambert	Female	42	45	17/06/2009
R151303	Cape Lambert		34	45	17/07/2002
R163490	Cape Lambert	Male	40	20	10/07/2008
R163491	Cape Lambert	Female	40	58	10/07/2008
R163492	Cape Lambert	Sub Adult	31	20	10/07/2008
R163493	Cape Lambert	Female	40	48	10/07/2008
R163494	Cape Lambert	Male	39	32	10/07/2008
R164337	Cape Lambert	Male	39	40	6/10/2007
R164338	Cape Lambert	Male	34	51	6/10/2007
R164339	Cape Lambert	Male	38	55	6/10/2007
R164340	Cape Lambert	Male	39	31	7/10/2007
R164341	Cape Lambert	Male	36	33	6/10/2007
R164342	Cape Lambert	Male	38	48	7/10/2007
R164343	Cape Lambert	Male	38	47	06/10/2007
R166911	Dixon Island	Female	34	48	13/01/2009
R166912	Dixon Island	Juvenile	25	34	14/01/2009

suitable habitat is present but no specimens have been recorded during targeted searches in that area. However the habitat in the Karratha and Cleaverville Beach area is severely degraded by off road vehicles and sand mining operations.

On the mainland, the species has been recorded as far north as half way between Point Samson and Cossack on an isolated sand dune. However, no specimens have been recorded at Cossack or Settlers Beach which has a large expanse of suitable habitat.

Three other species of *Lerista* are known to occur in the region and are sometimes in sympatry with *L. neviniae*. These are *L. bipes*, *L. clara* and *L. verhmens*.

L. bipes is the most often-caught species of *Lerista* in the region, with 117 individuals recorded during one survey effort (GHD 2011). This species appears to utilise a range of habitat types and is sympatric with *L. neviniae*.

Lerista clara is recorded regularly in the Cossack, Karratha and Settlers Beach areas and, like *L. neviniae*, is found in loose soils under woody debris. *L. clara* is a member of the “muelleri” species complex and appears less common in areas where *L. neviniae* is present. Alternatively *L. neviniae* is absent where *L. clara* occurs in high numbers. Biota (2008b) found clear separation between *L. clara* and *L. neviniae* in searches and suggest that the presence of *L. clara* may limit *L. neviniae*’s presence.

L. clara is a slightly larger species than *L. neviniae* (SVL 49 mm versus 41 mm, respectively), appears to be more robust in habitat usage and occur over a much larger environmental range. It is possible that as *L. clara* moves into *L. neviniae* habitat the species is ‘out muscling’ its smaller counterpart. This could already be happening in the Point Samson area as several hand searches for *L. neviniae* in this area have been undertaken with only two individuals discovered near the town site. In prime *L. neviniae* habitat the species is relatively common and is generally found within the first 20 minutes of searching, with up to 25 individuals found at one site (Biota 2008b).

L. verhmens has been found living in the same area as

L. neviniae but is separated by its environmental selection. It generally favours a rocky, heavier clayey-loam soil, although at Cleaverville *L. verhmens* was recorded on coastal sands (Biota 2008b) and therefore was potentially sympatric with *L. neviniae*. On Dixon Island, *L. verhmens* was only recorded only on rocky ridgelines and *L. neviniae* in coastal dunes.

Impacts to the species

Lerista neviniae habitat lies within an area of Western Australia that is recognised as a key mining region with established port infrastructure and accommodation for workers. This area of the Pilbara has experienced considerable interest for future developments in mining and infrastructure expansions which could impact on *L. neviniae* habitat. Currently the species is known to occur in approximately 600 ha of coastal habitat. With on-going development plans in the area this is likely to reduce the amount of habitat available for the species.

The human population of the region has increased in response to the mining industry; as such the pressure on recreational areas in the region has increased off-road vehicle traffic and pedestrian movement on coastal dune and beach habitats. This is evident at Cleaverville Beach and numerous small sandy beaches near Point Samson. Vehicle traffic on dunes is likely to affect the stability of the dune and the vegetation. Additionally, fires and fire-wood collection in these areas has reduced fallen woody debris for the species to hide, limiting micro habitats available. This in turn may increase competition with *L. clara*, although this needs to be further investigated.

During hand-foraging on Dixon Island in 2010 that targeting dead *Cenchrus ciliaris* (Buffel Grass) clumps, it was noted that at least two dead *L. bipes* were entangled in buffel grass root systems. Both specimens were petrified but appeared stuck, woven between the upper woody stalks of the remaining roots. No dead *L. neviniae* were recorded, however, Buffel Grass is commonly found in *L. neviniae* habitat and may impact on populations; however, this is speculative and needs to be further investigated.

CONCLUSION

L. neviniae was thought restricted to the Cape Lambert area; this study demonstrates a slightly broader distribution although the extent of suitable environment is only approximately 600 ha. With additional surveys new areas may be identified but available habitat for this species is limited to coastal or near coastal fringe pale dunes in the western Pilbara. The species' range is within an area that is experiencing expansion via mining and infrastructure projects and may impact on the species long term persistence. Therefore, while currently listed as vulnerable, it may meet a higher category due to the area of occupancy, population size and likely threats. Additionally none of the species' range is currently protected under conservation estate.

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Oral delivery of double-stranded RNA and the effects of RNA interference on the green peach aphid, *Myzus persicae*

VINEETA BILGI*, JOHN FOSU-NYARKO & MICHAEL G. K. JONES

Plant Biotechnology Research Group, WA State Agricultural Biotechnology Centre,
School of Veterinary & Life Sciences, Murdoch University, Perth, Western Australia, 6150

* Corresponding author: ✉ vineetabilgi@gmail.com

The green peach aphid (GPA), *Myzus persicae*, is a polyphagous insect that feeds on a broad range of hosts. It also transmits over 100 plant viruses. As a result, feeding damage and the viral diseases the aphid transmits contribute to yield losses in several economically important crops worldwide including potato, canola and lupin. Genetic resistance and application of insecticides are currently the primary means of controlling GPA. However, insecticides are not always effective: for example, carbamate or neonicotinoids are chemicals to which GPA can readily develop resistance. An alternative approach is the use of the naturally occurring phenomenon of RNA interference (RNAi) to silence essential genes in the aphids. RNAi is the sequence-specific degradation of homologous RNA molecules guided by small RNAs, and can be triggered by the introduction of double-stranded RNA (dsRNA). This approach is potentially economical, environmentally-safe and has been shown to be an effective strategy against several different plant pathogens and pests including nematodes and some insects.

Several methods have been developed to deliver dsRNA to insects such as soaking cell lines, microinjection, oral feeding (*ad libitum*) and *via* transgenic plants. Recently, genes of insect pests of the orders Lepidoptera, Coleoptera and Hemiptera have been silenced. For example gene silencing in the western corn rootworm (*Diabrotica virgifera virgifera*), the cotton bollworm (*Helicoverpa armigera*), the pea aphid (*Acyrtosiphon pisum*), and GPA through injection, feeding artificial diets containing dsRNA (*in vitro*) or using transgenic plants expressing dsRNA (*in planta*) has been attempted. Microinjection of dsRNA into the abdomen or thorax involves the use of very fine glass needles to administer dsRNA into various nymphal stages of insects. This method requires use of appropriate size of needles and substantial skill to reduce undue stress, injury or even death as well as empirical injection of volumes and doses of dsRNA. Microinjection can be laborious and may not be suitable where a large number of insects are required in a large-scale functional analysis (of target genes) and for small nymphs of some insects, as they may be prone to injury and stress.

Oral delivery of dsRNA to insects (also called artificial feeding) involves administration of dsRNA suspended in

sucrose solution in a feeding chamber: artificial feed that is sandwiched between thinly stretched parafilm layers is placed over the mouth of the container and aphids are allowed to feed. This method is convenient, non-invasive, allows the use of nymphs; *ad libitum* feeding is close to the natural feeding behaviour of aphids. Artificial feeding is also well suited for large-scale functional analyses and has also been used successfully to screen 290 dsRNA targets in the western corn rootworm. One of the limitations of oral delivery is that it is difficult to ascertain whether the insect has ingested dsRNA and the amount ingested. However, most published artificial feeding systems for RNAi in insects do not use markers to trace uptake, and thus all experimental insects are assumed to have fed, and the effects of gene silencing are averaged. This could result in underestimation of gene knockdown. However, if uptake of solutions can be traced, only insects that have taken up 'feed' containing dsRNA will be identified and used for accurate analysis of gene silencing. For example, fluorescent-labelled Cy-3 has been used to investigate uptake of dsRNA in Hemipterans such as the glassy-winged sharpshooter, the grain aphid and the potato/tomato psyllid. However, Cy-3 labelling is expensive when replicated experiments for testing many different target dsRNA are involved. The use of inexpensive vital dyes to trace uptake of dsRNA may allow effective assessment of aphids in such experiments.

The overall aim of the current study were to investigate the effects of *in vitro* RNAi in GPA through oral delivery of dsRNA. The objectives involved: identification of dyes that were non-toxic to GPA, did not affect the stability of dsRNA, and could be easily seen inside the aphid body as an indicator to uptake of dsRNA solutions. For this, suitability of 11 inexpensive dyes were studied. Effectiveness of this system was assessed by studying the effects of silencing a proton-translocation gene. To do this, GPA nymphs were fed with the target dsRNA for 24 hrs *in vitro*, after which phenotypic effects and percentage of active aphids with dyes were assessed as well as presence of dyes in the aphid body. Transcript abundance was measured through semi-quantitative PCRs after 24 hrs of feeding on dsRNA. To assess long-term effects of feeding on survival and fecundity of GPA, aphids fed on dsRNA with visible dye and those fed without dye were transferred onto tobacco and monitored daily for 12 days.

Of the 11 dyes tested, optimal concentrations of two were effective in tracing uptake of feed/dsRNA and allowed the efficient assessment of RNAi in GPA. The vital dyes neutral red (NR) and acridine orange (AO) were easily seen in the salivary glands and alimentary tract of aphids at concentrations of 0.02% and 0.0025% respectively. Based on spectrophotometric analysis and

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agarose gel electrophoresis, the quality and stability of dsRNA was not affected by NR or AO. Our results also indicated that 2 µg/µL of dsRNA of the target gene was effective in inducing RNAi in GPA. After feeding for 24 hrs, there was 22% reduction in survival of aphids fed on dsRNA with NR and 30% reduction in those fed on dsRNA without NR as compared to controls. In both cases, there was a significant reduction in transcript levels as compared to dsGFP and no-dsRNA controls ($p<0.05$) and expression of the target gene was not affected in aphids fed on dsGFP. Interestingly, transcript abundance of the target gene in the aphids fed on dsRNA with dye was much lower than in those fed without dye, indicating a more pronounced gene knockdown in the former. This demonstrates that assessing gene expression in only those aphids that have fed provides a better measure of

transcript abundance than in aphids pooled together. There were also long-term effects on aphid survival; the treated aphids remained alive on tobacco for only five to six days as compared to controls indicating silencing this gene disrupted the normal life-cycle.

To our knowledge, this is the first study that makes use of vital dyes to trace the uptake of dsRNA through *in vitro* RNAi in GPA. This study suggests that the use of vital dyes as internal markers in feeding-based RNAi has great potential and allows assessment of only those individuals that have taken-up the feed. Both NR and AO are easy to use and inexpensive compared to the fluorescently labelled Cy-3 dye and will provide an economical option in RNAi studies. The optimised concentrations of NR and dsRNA are being employed successfully in RNAi of GPA genes *in vitro*.

Biogas for Sub-Saharan Africa: Current situation and opportunities for improving dissemination

GLORIA V. RUPF*, PARISA ARABZADEH BAHRI, KARNE DE BOER & MARK P. MCHENRY

School of Engineering and Information Technology, Murdoch University, 90 South Street, Murdoch, WA, 6150

* Corresponding author ✉ g.rupf@murdoch.edu.au

Biogas technology has the potential to improve energy access and waste management practices in both urban and rural Sub-Saharan Africa (SSA), along with making a positive contribution to health and the environment. The last seven years have seen a significant increase in the uptake of domestic biogas systems in a selected number of SSA countries through the Africa Biogas Partnership Programme (ABPP). Before ABPP, biogas dissemination has been slow and sporadic in the region, and even today much of the potential is still untapped. One barrier has been the focus on a selected few biogas systems that use cattle dung as their main feedstock. This paper will provide key recommendations on how biogas dissemination in SSA can be improved, and highlight the energy production potential from a range of feedstocks. A proposed biogas system model will be introduced which could become a valuable tool for identifying optimal biogas system designs for particular applications in SSA.

The livelihoods of millions of households in Sub-Saharan Africa could be significantly improved by biogas technology providing an alternative fuel for cooking and lighting. The majority of SSA households, particularly in rural regions, currently rely on traditional biomass sources including firewood, charcoal, cattle dung, and crop residues for cooking. Traditional cooking stoves are inefficient and produce indoor pollution, which causes a number of lung, and some heart diseases. Furthermore, the collection of firewood contributes to deforestation, land degradation, aggravated soil erosion, and flooding. Biomass collection can also be dangerous for women and children who often carry out the work, and takes away time from other important activities, including attending school. Biogas technology harnesses the anaerobic digestion process to convert organic waste into biogas and a nutrient rich effluent that is suitable as fertiliser. Biogas is a mixture of 50–70% methane, 30–45% carbon dioxide, as well as containing other trace gases, which can be used like natural gas for cooking, heating, electricity, and as a vehicle fuel. Biogas use in SSA not only provides an alternative cooking fuel, but also provides fertiliser and an effective method of treating organic waste. For example, household biogas systems are amenable of the connection of a latrine for improved sanitation.

Dissemination of biogas technology in SSA has been sporadic since its first introduction in the 1950s. Domestic biogas programmes operating under ABPP

are currently limited to Burkina Faso, Ethiopia, Kenya, Tanzania and Uganda. The most common type of biogas system used in these programmes are fixed dome digesters, with some plug flow and floating cover digesters also being installed. The main barriers hindering larger dissemination include high installation costs, inadequate user training, insufficient servicing, and inappropriate designs. Poor design choices, mainly due to overlooking the user energy needs and local conditions, contribute to the short lifespan of many installed biogas systems. The key recommendation for improving biogas dissemination is improving the design choices of biogas systems. This requires modifications to be made to existing system designs to suit the context and needs of the intended user. Technical considerations such as the surrounding environmental conditions, technical skills, and materials available, along with the socio-cultural context and needs of the user, need to be included in the system design. Experience in parts of SSA and other developing regions has shown that collaboration between research institutions, governmental departments and potential as well as current biogas users helps increase its dissemination. Such collaboration can include establishing knowledge sharing hubs for biogas technology to ensure the technology continues to be developed and applied more efficiently and appropriately.

The total estimated methane production potential from suitable biogas feedstocks in SSA is 23.7 billion m³/year and 245,600 GWh of energy. Suitable biogas feedstocks include livestock manure, livestock food waste (eggs, hides and skin, and milk), crop waste normally burnt, crop equivalent food waste, domestic sewage, and the organic fraction of municipal solid waste (MSW). Crop waste normally burnt and crop equivalent food waste make up the majority of the potential, with 43% and 26% of the total, respectively. Given that current domestic biogas programmes in SSA focus on cattle manure as the main feedstock, this highlights the opportunity to improve waste management practices through harnessing the other abundant organic waste streams in biogas systems. It is therefore recommended that biogas dissemination programmes in SSA extend their scope to include a wider variety of feedstocks, particularly food and crop wastes.

This research aims to address the gap in appropriate designs of biogas systems through the development of a biogas system design model that identifies optimal designs for particular applications in SSA based on user defined priorities of sustainability criteria. Development of the biogas system design model requires identification of the essential parameters that influence biogas production and digester design, a review of biogas technologies applicable to SSA, assessment of feedstocks

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available and selection of appropriate sustainability criteria. The parameters included in the model and considered essential in the design of biogas systems are: fertiliser and energy requirements; feedstock (type, amount, and rate of supply); water supply; land area available for installation; climate (ambient temperature and rainfall) and the construction materials available locally. Six main types of biogas digesters have been identified for the model, including batch reactor, continuously stirred tank reactor (CSTR), fixed dome digester, fixed film digester, floating cover digester, and plug flow digester. A wide range of feedstocks are considered, including crop residues, livestock manure,

MSW, food waste, and domestic wastewater. The sustainability criteria that can be ranked by the user according to priority in the model are environmental impact, cost, social contribution, ease of operation, reliability and robustness, as well as technical efficiency. Multi-criteria decision-making methods are applied in the model to identify the optimal biogas system design based on the priority rating of the criteria and the system designs that are feasible according to the main parameters. The biogas system design model presented in this paper empowers potential biogas users in SSA to realise the energy and biogas technologies that could be available to them to treat their organic wastes.

Stratigraphic architecture and evolution of a barrier seagrass bank in the mid-late Holocene, Shark Bay, Western Australia

GIADA BUFARALE & LINDSAY B COLLINS (DECEASED)

Department of Applied Geology, Curtin University, Bentley, WA 6102

✉ giada.bufarale@postgrad.curtin.edu.au

Shark Bay is located approximately 800 km north of Perth, Western Australia, and was awarded World Heritage Status in 1991. One of the most important geological features in Shark Bay is the Faure Sill complex, which is a well-developed carbonate-clastic barrier bank that lies east-west across the axis of the Hamelin Pool and L'Haridon Bight. Faure Sill has had a critical role in the evolution of the local ecosystems, in particular in restricting the water exchange between the southern embayments and Hopeless Reach. These environmental settings, along with a semi-arid to arid climate, have produced and preserved the metahaline and hypersaline conditions in the Hamelin and L'Haridon basins, providing a basis for the development of a variety of biogenic and physical structures such as microbial communities (stromatolites) and oolitic shoals that proliferate in the southern basins.

A combination of remote sensing images, high-resolution shallow seismic data and lithostratigraphic information provided new insights into the internal architecture, sediment bodies, lithofacies and Holocene development of the Faure Sill. By integrating these data with radiocarbon dating, information concerning the sediment accumulation rates was obtained, together with an estimated chronology, bank onset and growth history.

An initial remote sensing analysis of high-resolution aerial photos of the Faure Sill allowed mapping of the sediments and benthic substrates and the morphological elements, demonstrating the interconnection between channel – fan – levee – bar morphology.

A high-resolution shallow geophysical survey was successively carried out using a subbottom profiler. The seismic profiles provided useful information on the thickness, lateral distribution and acoustic characteristics of the sediment packages and geometry of the seismic reflectors, giving good quality data on the Holocene and Pleistocene morphostratigraphic features, down to a depth of about 15 m below the seafloor.

Key acoustic reflectors were identified and, in order to integrate and assess their nature and occurrence, a follow-up coring study was performed. A total of 32 cores, up to 6 m long, was collected with a vibracorer, along the seismic lines. The sediment cores were then logged and the composition and mineralogy of the lithofacies were analysed. Samples from four cores were

collected for radiocarbon dating in order to calculate accumulation rates and assess the age of reflectors and facies. With this information, it was also possible to estimate the Mid-Late Holocene history of bank and create a local stratigraphic model of the seagrass bank growth.

The results indicate that:

- Acoustic reflectors in the seismic profiles coincide with an abrupt variation of sediment types, which is evident from the sediment core analyses, and only minor differences in terms of position and frequency occur.
- The Faure Sill is composed of three different types of sediment: sand, mud and clay. Bioclastic muddy sediments are by far the most common bank lithofacies. Sands are predominately bioclastic and the particle size ranges from coarse (generally cross bedded) to fine (mainly thinly laminated). Seagrass peats are also a significant component. Bioclastic, muddy facies are associated with lower parts of the seagrass bank buildup and are succeeded by sandy facies. Fringing banks associated with the Wooramel Delta are muddy and contain a significant component of terrigenous clays.
- On the basis of internal architecture, facies and distribution and morphological features, the sediment bodies can be classified into five main types: 1) Channel-associated; 2) Tidal fan-associated; 3) Banks; 4) Bank top; and 5) Shoreline-attached sand lobes.
- With radiocarbon dating, information about the accumulation rates was obtained. A number of hiatuses were recognised and correlated to seismic and lithological boundaries, and these data assisted in the development of bank accumulation curves. Accumulation rates varied in time and magnitude, in accordance with seagrass productivity, local hydrodynamic conditions, water depth and facies. The average accumulation rate, considering all facies of the Faure Sill, is 1.3 m/ka, ranging from 0.7 m/ka in the eastern part, adjacent to the Wooramel Delta, to 1.7 m/ka in the central and western sector of the carbonate bank. Within the cut-and-fill channel, bank accumulation rates significantly accelerated, peaking at 6.9 m/ka.

The study demonstrated that the evolution of the marine ecosystems of Shark Bay resulted from an interconnection between sediment body morphologies, seagrass, which produced and trapped a large amount of sediments, and pre-existing topography that shaped the initial deposition. Sea level fluctuations have also

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largely controlled the hydrodynamic conditions, such as the amount of flushing waters and their velocity, influencing the erosion, transportation and deposition of the sediments.

Combining the data, a model of the Holocene chronology of the Faure Sill was generated. The development of the bank can be seen in three sea level stages.

- 1) *Early Holocene*. As the sea level has risen (not earlier than 8000 years BP), a pre-Holocene irregular topography was flooded. During this period, the first seagrass communities started their colonisation, contributing to the initiation of an early bank, trapping quartz, shed from eroding pre-existing topography, and producing *in situ* bioclastic sediments and muddy carbonate deposits.

- 2) *Middle Holocene*. The sea level peaked (about 2.5 m above the present sea level), completely flooding the bank region; under these conditions, the seagrass meadows reached their apex and muddy sediment accumulation peaked around 6800 years BP, corresponding to the sea level maximum highstand.

- 3) *Late Holocene*. As the sea level dropped in the period up until the present time, several distinct erosional and depositional events took place, and sandy bank top facies developed. During this time, the deposits continued to fill the available accommodation space and channel bank morphology continued to develop.

A fuller account of this work can be found in: Bufarale, G and Collins, L B, 2015, Stratigraphic architecture and evolution of a barrier seagrass bank in the mid-late Holocene, Shark Bay, Australia, *Marine Geology*, **359**: 1–21.

Stabilized frequency transfer for the Square Kilometre Array

D. GOZZARD¹ & S. SCHEDIWY

School of Physics, The University of Western Australia

¹Corresponding author ✉ david.gozzard@research.uwa.edu.au

The Square Kilometre Array (SKA), due to begin construction in 2018, will be the largest radio-telescope ever constructed, and will incorporate clusters of antennas placed up to 150 km from the core site. Phase-coherent frequency reference signals must be distributed to each antenna in the SKA network in order for the array to function properly. The conventional method for a telescope of this extent would be to install individual atomic clocks (particularly hydrogen-masers), housed in temperature-controlled, vibration-isolated rooms, at each antenna cluster. This is extremely expensive to both construct and maintain and the SKA Office intends to employ stabilized signal dissemination systems as a much more cost-effective way of delivering coherent reference signals to each antenna over such large distances.

The stabilized transfer system developed in the School of Physics at the University of Western Australia uses a continuous-wave laser that is modulated at the desired reference frequency and transmitted via fibre link to the receiver. A portion of the signal is reflected from the receiver unit back along the same fibre link to the transmitter, where it is compared to the original reference signal. The disturbances on the transmission line can then be measured and the out-going signal adjusted in order to account for these disturbances.

The modulation of the continuous-wave laser is achieved by splitting the optical signal from the laser into the two arms of a Mach-Zehnder Interferometer (MZI). Acousto-optic modulators (AOMs) in each arm of the MZI apply separate frequency shifts to the split optical signals. The shifted optical signals are recombined at the output of the MZI and are sent via fibre-optic cable to the receiver at the antenna. The difference in frequency between the two, shifted optical signals is the desired radio-frequency reference for use at the antenna. The two optical signals arriving at the antenna are frequency shifted again by an AOM in the receiver and a portion of their power is reflected back down the fibre link to the transmitter. The frequency shift at the receiver allows the transmitter to distinguish signals returning from the receiver from spurious reflections due to damage on the transmission line. The returning signals are beat against the original signal from the continuous-wave laser (that is, the returning signals are compared to the original laser signal) producing two radio-frequency beat signals that carry information about the frequency shifts caused by

both the AOMs and the disturbances on the transmission line. Further electronic mixing of these beat signals reduces them to a form suitable to steer a PI controller, which applies a frequency modulation to one of the two transmitter AOMs calculated to pre-compensate the disturbances on the link.

A key feature of this system is that it uses optical detection (i.e. the optical heterodyne beat) to govern the modulation of a radio-frequency transmission. Most similar systems detect link disturbances in the radio- or microwave-frequency domain. Detection in the optical domain has advantages in sensitivity (due to the much shorter wavelength) and it is relatively easy to shield from spurious reflections from the fibre link using the frequency shift applied by the receiver AOM.

The SKA telescope is split between the African and Australian continents. Due to the geology of the South African SKA site, it is extremely expensive to bury the fibre transmission lines to the antennas and it would be much simpler and cheaper to run the fibres along the over-head power lines also going to the antennas, but this makes the fibre lines extremely susceptible to disturbances caused by the wind. Using our optical-heterodyne detection system, we measured the signal instabilities on a 150 km link at the South African SKA site to be approximately 3500 times greater than on a buried 1840 km link in Germany measured by Droste and colleagues in 2013. Despite this unprecedented level of noise, our stabilization system was able to effectively suppress the instabilities on a 20 MHz transmission, even at wind speeds higher than the maximum operating requirement of the SKA telescope, and achieved a phase stability of 0.01 radians at an integration time of one second. Based on these results and previous laboratory tests of similar signal stabilization systems operating at other frequencies, it is likely that a similar level of stability will be achieved by systems designed to transmit the 350 MHz and 32 GHz reference frequencies required by SKA-low and SKA-mid. That is, these systems are expected to achieve a signal stability around 20 times lower than the SKA requirement of 0.2 radians at one second integration time. Thus the use of stabilized frequency dissemination technologies such as ours will enable the South African SKA site to use over-head transmission lines, saving the project millions of euros that it would cost to bury the fibre.

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